Nitrate Pollution Control in the Presence of River Flow Restrictions and Weather Variability: The study of a Scottish Agricultural Catchment

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Thesis submitted in candidacy for the Degree of Doctor of Philosophy

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December 2002

Contents

Abstract

Both low river flows from surface water extraction for irrigation and diffuse nitrogen pollution are agricultural externalities. Efficient environmental regulation at the catchment level requires that the two be considered together. An economic analysis of policies to control non-point source nitrogen pollution in the presence of minimum river flow controls in a Scottish agricultural catchment was undertaken. A realistic nonlinear Bio-Physical Economic model was constructed which related farming activities (crop/soil land allocation, nitrogen fertiliser application, livestock husbandry, surface water extraction for irrigation etc.) with catchment profitability and environmental externalities (low river flow and diffuse nitrogen pollution). Numerous economic, managerial and mixed regulatory policies were ranked in terms of overall reduction in welfare arising from loss of profitability under regulation.

The presence of minimum river flow controls in the catchment was found to reduce nitrogen pollution. This reduction was sufficient to be considered in the design of diffuse pollution policies. However river flow controls did not, for the most part, alter the relative ranking of instruments. By themselves, river flow controls were found not to be a cost effective means to reduce diffuse nitrogen pollution.

The effect of varying weather patterns on the relative ranking of polices and the levels required to meet the standard was considered. Although the overall efficiency of economic controls targeting emissions was established, mixed instrument policies did particularly well in 'wet' weather conditions, while economic controls targeting nitrogen as an input performed poorly in the representative 'wet' weather conditions.

Acknowledgements

This thesis would not have been possible without the advice, encouragement and help of many people. Firstly I would like to thank my supervisors Prof. Nick Hanley, Dr. Athanasis Kampas and Dr. David Ogelthorpe for their guidance and patience. I am particularly grateful to Prof. Hanley for being an exemplary mentor and his indisputable support over the years. I must specifically recognise Dr. Kampas' contribution and above all his inspirational tutelage which fostered my enthusiasm for research.

I must thank the Macaulay Land Use Research Institute (MLURI) for organising, financing and supporting this research in every possible way; without the Macaulay Development Trust this thesis would not have been possible. I appreciate the contribution made by Dr. Sarah Dunn, Dr. Bob Crabtree, Dr. Mark Stalham, Wendy Van Bremium, Carol Smith, Andrew Thorburn, Peter Shannon and Neil Chalmers at various stages of the research. I am grateful to Dr. Mark Stalham, Dr. Allan Lilly and Dr. Eunice Lord for their contribution to the scientific modelling and being patient with an economist.

I am thankful to Channi for being there and making the Ph.D. bearable. Above all I am indebted to my parents and Iza for their continual encouragement and invaluable guidance.

Glossary of Terms and Acronyms

CAP	Common Agricultural Policy
EU	European Union
NSA	Nitrate Sensitive Area
NVZ	Nitrate Vulnerable Zone
NPS	Nonpoint Source Pollution
WFD	Water Framework Directive

Chapter 1
Introduction

1.1 Chapter Outline

This chapter is divided into three parts. The first briefly introduces the regulatory problem, its policy relevance and research objectives. This is followed by an explanation of the evolution of environmental policy in the European Union (EU) from the perspective of the Ecological Modernisation Theory. The last segment of the chapter presents an abridged account of the thesis.

1.2 The Research Objectives

The broad remit of this research was to study the use of 'integrated catchment management' and 'full cost recovery pricing' in implementing the EU Water Framework Directive (WFD) (E.U. Sep 2000). The research focuses on non-point pollution control and water abstraction, and models a Scottish catchment as a case study. The WFD is the most comprehensive re-working of European legislation on water regulation to date, and supersedes/incorporates previous EU water-related Directives. In terms of policy implications, three of the directive's main requirements are:

1. The requirement *to achieve good ecological status* (discussed later) in all water bodies i.e. lakes, rivers, estuaries, coasts etc., unless derogations are sought and granted, throughout the EU.

2. That '*full cost recovery pricing*', be implemented in principle as prioritised in the 5th Environmental Action Programme. This implies evaluating and accounting for '*environmental and resource costs*' (Article 9). Thus agricultural *externalities* such as reduced river flows and diffuse nitrogen, phosphorus and pesticide pollution etc. should be priced and internalised accordingly. Adequate water pricing provides an incentive for the sustainable use of water resources and thus helps attain environmental objectives (Europa 2002).

3. The requirement of '*integrated water catchment management*', which stipulates the integrated organisation and regulation of water management at the 'catchment' or river basin level. The WFD requires that all river basins and coastal waters must be assigned to a River Basin District (RBD). Integrated management implies the joint management of water quality and water quantity, for instance in the management of polluting inputs *and* water abstraction.

While the UK is likely to be covered by approximately 14 River Basin Districts, each to become a single planning unit, much of the detailed catchment planning will occur at the Sub-Basin scale. Sub-Basins will be defined according to need, and so it can be reasonably expected that more detailed planning (smaller sub-basins, or groups of sub-basins) will take place in areas of greater water stress. The East of Scotland is one such area, where rainfall is naturally low, and where the demand for irrigation water regularly exceeds availability, resulting in problems of both water availability and water quality.

Low flow rates in rivers and burns, exacerbated by abstraction, are a predicted cause of failure (Scottish Executive, 2002). There is evidence to support the need for further surface water extraction controls in intensively irrigated Scottish catchments (Fox 1999; MLURI 2001).

The European Environment Agency has recently corroborated that diffuse nutrient (nitrate, phosphate) loads from agriculture are responsible for eutrophication of coastal and surface waters and nitrate contamination of aquifers (EEA 2002). Nitrate levels are seen as an important reason for the failure of some Scottish surface and ground waters to reach "good status" as specified under the WFD (SEPA, 1999). Diffuse pollution from nitrogen, which results in eutrophication, contamination of potable water supply and acidification has been widely recognised and partially addressed in Scotland (SEPA 1999; Darcy et al. 2000).

This research empirically investigated: a) the impact of river flow restrictions on agricultural non-point nitrogen pollution control; b) compared the relative efficiency of policies to control diffuse nitrogen pollution based on mean and wet weather conditions; and c) considered mixed instrument policies which may be more appealing to regulators. In addition, the theoretical internalisation of two surface water agricultural externalities, i.e. nonpoint source nitrogen pollution and reduced river flows from surface water irrigation, is presented. The dual nature of surface water diffuse nitrogen pollution as both a *positive production* and *negative environmental externality* is considered. The West Peffer catchment (Scotland) was modelled due to the prevailing reliance of farming on surface water irrigation and intensive cultivation.

Although there is an abundance of literature on nonpoint pollution (Dosi and Tomasi 1994b; Xepapadeas 1997; Shortle and Horan 2001) and some investigating the use of irrigation controls to control diffuse pollution (Pfeiffer and Whittlesey 1978; Stevens 1988; Dinar and Letey 1991; Weinberg et al. 1993; Booker and Young 1994; Zerki and Herruzo 1994; Helfand 1995; Larson et al. 1996; Murillo et al. 2001) there is no study to my knowledge, which empirically investigates the effect of imposing minimum river flow restrictions on the control of catchment nitrogen pollution.

Nitrate emissions from farming are a damaging bi-product (exposure may result in eutrophication, 'blue baby syndrome or various forms of cancer[1]) or externality of intensive agricultural production. Although farmers have a private incentive to apply as much nitrogen as is profitable (nitrate emissions have a positive shadow value to farming (Chambers and Quiggin 2000) the rest of society would prefer to restrict their usage below the privately optimal level.

Bio-physical economic simulation modelling can, to an extent, a) overcome the information asymmetry between the principal/regulator and agent/farmer and b) the regulatory inability to observe agricultural pollutant run-off (Weersink et al. 1998), thereby converting a non-point pollutant problem into a point one (Shortle and Abler

[1] Chapter 2 discusses the environmental and human impacts of nitrate exposure

1997). However, the issue of estimating the external cost of agricultural externalities remains contentious and problematic (Xepapadeas 1997) therefore policy relies on the use of second-best environmental standards (e.g. maximum ambient pollutant concentrations and minimum river flow restrictions).

This research uses bio-physical economic modelling to assess the relative performance of second-best policies to regulate diffuse nitrate pollution at the catchment level in the presence of minimum river flow restrictions. A challenge facing regulation is to implement the Nitrate Directive (91/616) which sets an upper bound (standard) of 11.3 mg nitrogen per litre (or 50 mg/litre nitrate) in drinking water - at least cost to catchment farming, given the catchment has to meet specific minimum river flow requirements.

A theoretical hypothesis is developed (chapter 5), which accounts for interaction between controls on water abstraction and controls on nutrient use. This interaction affects the cost-minimising outcome for achieving target reductions in ambient nitrate levels. Essentially, regulators obtain an extra degree of freedom by extending controls over water abstraction, rather than just nutrient inputs and land use. Introducing twin targets over minimum water flows *and* ambient nutrient levels (as is implicit in the WFD) makes the regulatory problem more interesting and complex. The empirical component of this research investigated this inter-action using a catchment level bio-physical simulation model.

1.3 EU Surface Water Ecological and Chemical Protection

In the European Union historically there has been a dichotomy in pollution control. One type of control concentrated at the source, through the application of technology; while the other focused on the receiving body's environmental impact, i.e. quality standards. Where there is a concentration of pollution sources, source controls can result in a cumulative build up of harmful pollution. Similarly quality standards can also underestimate the effect of a particular substance on the ecosystem, due to the limitations in scientific knowledge regarding dose-response relationships and the pollutant transport mechanism. The WFD integrates the two by requiring implementation of all existing *technology-driven* source based controls, besides implementing existing environmental quality standards (Europa 2002). The required end result is to ensure overall 'good' environmental status for all waters.

The WFD introduced a general requirement for ecological protection and a minimum ambient chemical standard to cover all surface waters. These were termed 'good ecological status' and 'good chemical status' respectively. Annex V of the Water Framework Proposal defines good ecological status in terms of the quality of the biological community as well as the hydrological and chemical characteristics. Due to the biological variability within the EU no absolute biological quality standards can be set. Instead controls have been specified in terms of conditions occurring if there was minimal anthropogenic impact. Procedures which allow identifying such conditions (chemical, hydromorphological) for a given body of water are detailed in the directive; including a system to ensure consistent compliance between member states. Good chemical status is ensured by compliance with numerous quality standards for chemical substances detailed in chapter 2. The directive also permits renewing these standards and establishing new ones.

The following chapter segment details the evolution of environmental policy in general. After this the Ecological Modernization Theory (EMT) is explained and related to the evolution of environmental policy in the EU.

1.4 Evolution of Environmental Policy

Neo-classical economics advocates that market failure (lack of clear property rights) is the primary source of environmental degradation and government intervention to overcome this market failure constitutes an appropriate response. Whereas political theorists contend that it is the combined effect of market failure and the government's inability to respond effectively (i.e. policy failure) that is to blame for environmental deterioration (Panayotou 1992).

Some argue that the convergence of interest between the bureaucracy and industry favours standardized solutions to environmental problems when they occur - even though more efficient proactive solutions are present. Standardised solutions ensure the regulatory (governmental) costs of policy design and implementation are reduced. Similarly to industry (polluters) such remedial controls are straight forward and relatively easy to accommodate. Consequently policy veers towards standardized solutions and not economically efficient, proactive, or innovative policies. Additionally it is believed that the 'invisible' impact of policies are often ignored (Gouldson and Murphy 1996). Thus it is technological, cultural, and institutional inertia which prevents the adoption of proactive environmental policies (Skou-Andersen 1994; Janicke and Weidner 1995).

It is argued that general environmental policy has not evolved from strategic thinking (proactive) but rather short term reactive approaches to ecological crisis or failure (Janicke and Weidner 1995). Popular typology views environmental policy passing through the following stages, a) primary stage: pollution issues avoided by moving either the source or receptor to separate cause and effect, b) secondary stage: dispersing pollution sources to ensure the effects become less apparent and/or externalised to the region surrounding the source, and c) tertiary stage: the installation of control technologies which contain at sources coupled with subsequent treatment (Skou-Andersen 1994).

1.5 Ecological Modernisation Theory (EMT)

EMT is both a normative and prescriptive theory which argues that 'economic growth and the resolution of ecological problems, can in principle, be reconciled' (Hajer 1996). It contends that the ecological crisis facing the world can only be resolved by further industrialisation (Spaargaren and Mol 1992). EMT regards environmental issues not as a crisis but as an opportunity; since is it assumes industrial innovation (technological change) in a market economy encouraged by an 'enabling state' will ensure environmental conservation (Blowers 1997). EMT promotes integration of environmental considerations in other policy areas and exploration of innovative policy measures. It also advocates 'cleaner' (not clean) technologies which improve *both environmental and economic performance*.

'It assumes that the existing political institutions can internalise ecological concerns or can at least give birth to new supranational forms of management...'(Hajer 1996).

As EMT does not require radical regulatory reforms nor criticise capitalism, it is widely hailed by governmental and organised industry (polluters).

1.6 EMT and the European Union

Some have argued that the European Union's statements of principle and intent match the four central themes of EMT, in that:

1) The European Parliament Resolution on Environmental Technologies (CEC 1980) acknowledges the possible contribution of the environmental technology industry to future employment and economic development. Similarly, the *EU White Paper on Growth, Competitiveness and Employment* (CEC 1993) links the environment with the economy and suggests resolving employment, environment and economic problems simultaneously.

2) The Maastricht Treaty specifically addresses the need to integrate environmental policy into other areas of policy. Whereas the 5th Environmental Action Programme (EAP) sets out further details on such integration.

3) The 5th EAP also advocates a broader range of innovative policy measures, such as information provision, education, voluntary measures, financial support etc. The above mentioned *White Paper* called for eco-taxation which would reduce tax on under-utilised labour resources and increasing tax on over-exploited environmental resources.

4) The same paper also called for promoting a 'cleaner technology base'. Similarly the *Communication on Economic Growth and the Environment* (CEC 1994) reiterates the need for research and development in cleaner efficient technologies.

Thus, it can be argued that the EU is following a version of the EMT, at least in theory if not in name (Gouldson and Murphy 1996).

The rest of this chapter presents a chapter by chapter outline of the thesis.

1.7 Thesis Outline

Chapter 2 consists of three main parts; the first explains basic facts about nitrogen the element, nitrogen fertilisers, and the soil nitrogen cycle with special emphasis on soil nitrogen losses. The second part lists the human and environmental effects of nitrogen water pollution, while the final segment details the policy response in the UK as a result of EU regulation. This includes a discussion on the nitrate sensitive area scheme (NSA), nitrate vulnerable zones (NVZ), the common agricultural policy (CAP) and its reforms, the water framework directive (WFD) and nitrate politics.

Chapter 3 details theoretical issues concerning NPS pollution control and its control through economic and non-economic regulation. The chapter begins with an explanation of the properties of NPS agricultural pollutants which make its regulation difficult. An explanation of first and second-best cost-effective economic solutions is presented followed by a detailed discussion of both performance (ambient and liability rules) and design (expected runoff, input and technology) based controls. The relative benefits and disadvantages of all are debated. Marketable pollution permits and the possibility of point-nonpoint trading are discussed separately. Other practical concerns in the control of NPS pollutants are considered, including the issue of property rights, the polluter pays principle and subsidies. The chapter concludes with an analysis of non-economic approaches such as education, and regulatory standards.

Chapter 4 reviews numerous empirical studies of NPS pollution control (nitrate, phosphorus, sediment and pesticides) including some that investigate non-economic controls. Cost-effectiveness of different control policies in each study is ranked, along with their modelling assumptions and analytical methodology. Conventionally, such studies are divided into aggregate (regional) and disaggregate (catchment or watershed). Particular emphasis is placed on research investigating: a) disaggregate catchment scale studies; b) the interaction between control policies targeting both water quality and quantity (surface and/or groundwater extraction); and c) the impact controlling one NPS pollutant may have on the generation of another NPS pollutant.

Chapter 5 examines the theoretical internalisation of both water quality and quantity externalities. The two externalities considered are diffuse nitrate pollution and low river flows from surface water extraction. It derives the optimal corrective taxes in a first-best and second-best world and investigates complimentary interaction between controls on both. Nonpoint source nitrate pollution is considered both as a negative environmental externality as well as a positive production externality to downstream surface water irrigators.

Chapter 6 introduces the West Peffer catchment (Scotland), which was modelled for the purposes of this research. The chapter states the characteristic catchment soil types, land use and rainfall distribution. It also explains how the crop production, nitrate leaching, animal husbandry, potato growth (irrigation), and economic modelling were performed. All associated data sources and modelling assumptions are listed.

Finally chapter 7 presents the results, the cost-effective ranking of the control instruments analysed and their possible policy implications. Chapter 8 states the main limitations of this study as well as possible directions for future research and improvement. The chapter concludes with a synopsis of the research undertaken and its main conclusions.

Chapter 2
Nitrate Pollution

2.1 Introduction

First, this chapter will outline the basic facts about nitrogen fertilisers and the soil nitrogen cycle, i.e. natural processes that add and remove nitrogen from the soil. This is followed by a discussion of the health hazards of nitrate pollution to the ecosystem i.e., humans, ecology and environment. Once the detrimental impact of diffuse nitrogen pollution is established the rest of the chapter discusses the European Union's (EU) policy response - with emphasis on the UK and in particular Scotland. Finally the chapter concludes with a brief discussion on the politics of nitrate pollution.

2.2 Nitrogen Facts

Nitrogen (L. *nitrum*, Gr. *Nitron*) was discovered by chemist and physician Daniel Rutherford in 1772. Nitrogen as a gas is colourless, odourless, and generally considered an inert element (named *azote*, meaning *without life*).

Carbon, Oxygen, Hydrogen and Nitrogen make up 96% of all living matter (Campbell 1993). Nitrogen is an essential component of amino-acids which are the building blocks of proteins (nucleic acids, chlorophyll, and enzymes). Without adequate nitrogen, plant growth is restricted and characteristic symptoms of nitrogen deficiency include yellowing and death of leaves, and stunted growth, and low yield (however too much can delay fruiting).

Although Nitrogen constitutes 80% of the Earth's atmosphere, it is chemically inert and cannot be used directly by most living organisms. N fixing bacteria through a unique process of nitrogen fixation can convert N from its gaseous form (N_2) into compounds suitable for uptake by living organisms. Under undisturbed natural conditions, inorganic N usually constitutes between 1 to 2% of total soil nitrogen. In an attempt to overcome this limiting factor to production, i.e. the slow biological fixation of N, man has resorted to chemical synthesis (requiring high temperature and pressure) which requires the annual expenditure of immense amounts of energy. It is estimated that about 40% of the world's protein needs are derived from atmospheric nitrogen fixed by the *Haber-Bosch process* and its successors, to produce ammonia (IFA 2002).

2.3 Nitrogen Fertilisers

As *Inorganic* nitrogen fertilizers are concentrated their transportation costs are considerably lower than bulky organic manures. Artificial fertilisers are often convenient to apply and usually the nitrogen contained in them is immediately available for plant uptake. However, some nitrogen fertilizers have been specially treated by the fertilizer manufacturer to slowly release nitrogen into the soil. These "slow-release" fertilizers reduce the need and cost of frequent fertilizer applications. Nitrogen fertilizers are also mixed with fertilizer materials containing phosphorus and potassium to produce a fertilizer blend. Common disadvantages of inorganic fertilizers are often associated with poor management such as applying too much. Over application of nitrogen fertilizer increases the risk of salt injury to the plant and the potential for the nitrogen from fertilizer leaching out of the root zone and polluting ground and surface water.

In comparison *organic* nitrogen fertilizers often have relatively low nitrogen content and are often required in larger quantities. They also contribute organic matter to the soil which improves the soil's physical, biological and chemical properties making it conducive to plant growth. In addition, many organic fertilizers contain all the plant nutrients needed for plant growth including micronutrients such as iron, zinc, and manganese. Organic fertilizers are often inexpensive and can draw upon locally-available waste materials such as animal manure or yard waste. Disadvantages of some organic fertilizers include odour, difficulties in transporting, applying and incorporating bulky organic materials, unsightliness, and the risk of nitrogen immobilization in materials having low nitrogen content. As an approximate rule of thumb, around 50% of the organic nitrogen applied to the soil will be available for plants to use the first season after application (Lord et al. 1999).

If farmers were to stop using fertilizer now, crop yields would drop significantly over a couple of years. Many countries which today use substantial amounts of fertilizers and export agricultural produce, would barely be in a position to feed their own population (Mengel 1992). The demand for fertilizer N cannot be completely covered by rotation with nitrogen-fixing leguminous species (such as clover or soybean) unless more than 40% of arable land was cultivated with leguminous species; thus, a substantial proportion of arable land would be used to produce N naturally. This would be more expensive than industrial production (Mengel 1992) and undoubtedly the price of agricultural produce would increase. Tables 2.1, 2.2, and 2.3 are figures on the world nitrogen fertilizer *production* (1997/98), world fertilizer consumption trend and world fertilizer *consumption* (1998/99) respectively (IFA 2002). In table 2.2 and 2.3 P_2O_5 and K_2O refer to phosphorus and potassium oxides respectively.

Table 2.1: Nitrogen fertilizer production, 1997/98

Region	Million Tonnes Nutrients
China	19.94
India	10.08
North America	14.97
West and Central Europe	**13.57**
FSU	7.99
Middle East	5.61
Others	12.77
World	**84.93**

Table 2.2: World Fertilizer Consumption Trend

Year	N	P_2O_5	K_2O	Total
	(million tonnes nutrients)			
1920/21	neg.	1.73	neg.	1.73
1930/31	1.30	2.77	1.39	5.46
1960/61	10.83	10.73	8.48	30.04
1970/71	31.75	21.11	16.29	69.15
1980/81	60.78	32.04	24.39	117.21
1990/91	77.56	36.07	24.61	138.24
1998/99	82.18	32.88	21.87	136.93

Table 2.3: World fertilizer consumption, 1998/99

Region	N	P_2O_5	K_2O	Total
	(million tonnes nutrients)			
Developing Asia	**42.77**	15.66	6.87	65.30
North America	**12.87**	4.52	4.81	22.20
Latin America	**4.71**	3.47	3.08	11.26
West Europe	**9.96**	3.50	4.11	17.57
Central Europe and FSU*	**4.71**	1.29	1.50	7.50
Others	**7.16**	4.44	1.50	13.10

**FSU : Former Soviet Union*

2.4 The Nitrogen Cycle

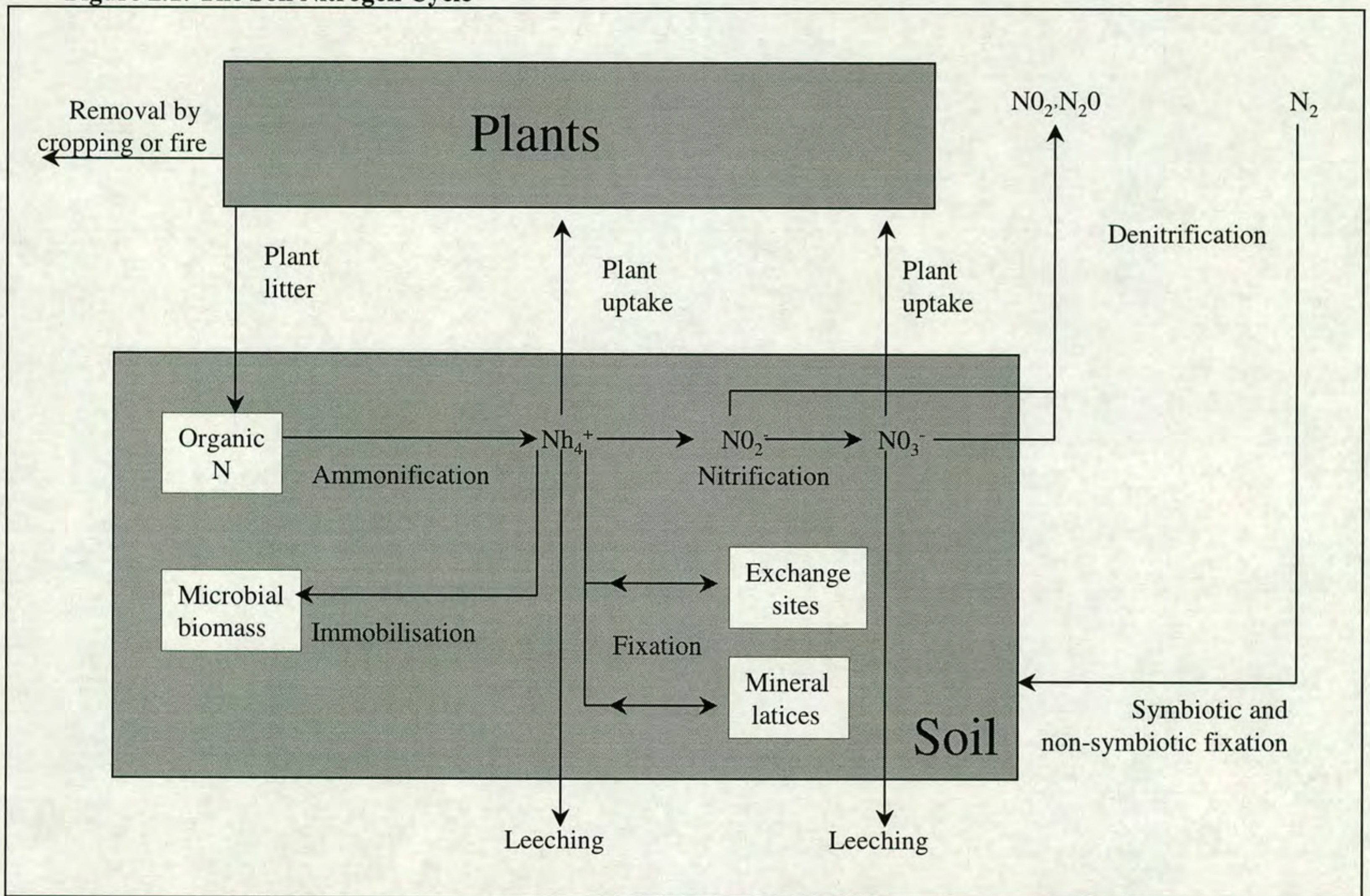
The nitrogen cycle in the soil can be broken up into those processes which contribute to the N available for crop uptake (soil nitrogen additions) and those which remove it from the soil (soil nitrogen losses). The soil nitrogen cycle is summarised diagrammatically in figure 2.1.

2.4.1 Soil Nitrogen Additions

Based on relative overall contribution, nitrogen inputs can be categorised into a) primary nitrogen sources such as inorganic (artificial fertilisers) and organic fertiliser (manure and crop residue), and b) secondary inputs such as atmospheric precipitation and biological fixation.

Nitrogen fixation: Nitrogen fixation is the conversion of dinitrogen (N_2) from the air to ammonium NH_4^+ and is carried out by nitrogen fixing bacteria which can be divided into two groups: a) non-symbiotic or free-living blue-green bacteria such as lichens and mosses; and b) symbiotic ones (genus *Rhizobium*) which live in the roots of leguminous plants (*Leguminosae*) such as peas, beans and clover where they form characteristic nodules. Healthy clover can fix more than 100kg N /ha/year. Biological fixation was central to maintenance of soil fertility by crop rotation, once widely practised in the UK (Hopkins 2000). Estimates of the amount of nitrogen fixation in the UK vary between 0.2 Mt and 0.4 Mt (The-Royal-Society 1983).

Figure 2.1: The Soil Nitrogen Cycle



Key

NO_2	Nitrogen Dioxide	N_2	Nitrogen (Gaseous)	NO_2^-	Nitrite Ion
N_2O	Dinitrogen Oxide	NH_4^+	Ammonium Ion	NO_3^-	Nitrate Ion

Atmospheric precipitation: Atmospheric precipitation of nitrogen (rainfall carrying the dry deposition of N to earth) and lightening discharges also add N to soil. It is estimated that up to 40kg N per hectare can come from the atmosphere. (Heathwaite et al. 1993).

Ammonification: The organic matter in soil consists of crop residue called 'humus' and microbial biomass which together make up the darker topsoil. The N content of plants (previous crops) and animals is decomposed by micro-organisms, which attack proteins and nucleic acids liberating ammonium (ammonification) (Pitcairn 1994). A part of N released is assimilated by the microbes themselves (immobilisation) – which in due course will be released when the microbe dies. However this process results in a net release of N and is referred to *mineralization*, because organic N is converted to inorganic form (NH_4^+ ammonium). Rates of ammonification depend on the size of the relevant microbial population, soil conditions, especially upon moisture content, temperature, acidity and the soil C: N ratio. Both ammonification and immobilisation take place simultaneously and which process dominates depends on soil characteristics and properties.

Nitrification: Two main mechanisms result in the microbial production of NO_3^- (nitrification). The common autotrophic nitrification involves oxidation of NH_4^+ to NO_3^- in two stages via NO_2^- is carried out by bacteria *Nitrosomonas* and *Nitrobacter.* The second mechanism is carried out by heterotrophic micro-organisms when organic N is transformed to NO_3^- . In general, nitrification is low in acidic soils, in nutrient poor soils, in soils with a high C: N ratio and /or low NH_4^+ availability and at low temperatures.

Both Ammonification and Nitrification are normally termed *soil transformation* however as they both contribute towards converting Nitrogen into forms which can be assimilated by plants they are have been grouped as 'additions'.

2.4.2 Soil Nitrogen Losses

Nitrogen losses from the soil come under the process of assimilation, denitrification, volatilisation, run-off and leaching.

Assimilation: Assimilation refers to the process of plant N uptake. Nitrogen is essential for life as it is necessary component of amino-acids which are the fundamental units of structural and enzymatic proteins. Plants are able to assimilate inorganic N (principally NH_4^+ and NO_3^-) and incorporate these into organic N compounds.

Denitrification: Denitrification is the microbiological conversion of NO_3^- to gaseous N such as, dinitrogen and nitrous oxide. It is the main process by which fixed N is returned to the atmosphere and is carried out by a host of organisms, but only under certain environmental conditions. It uses NO_3^- as a substrate and requires anaerobic conditions and a source of energy for the organisms. The amount of nitrous oxide emitted from soils globally is twice that amount produced by burning fossil fuels (Addiscott et al. 1991). The proportion of nitrous oxide to nitrogen gas emissions depends on soil properties. In general strongly acid soils usually emit only nitrous oxide while non-acidic soils in temperate regions mainly produce nitrogen gas unless there is a large amount of nitrates in the soil, as is the case after the application of fertiliser.

Volatilisation: Surface volatilization of N occurs when urea forms of N break down forms ammonia gases and where there is little soil water to absorb it. This condition occurs when urea is placed in the field but not in direct contact with the soil or when farm manure is allowed to decompose on the soil surface. The rate of surface volatilization depends on moisture level, temperature and the surface pH of the soil. The more alkaline a soil is and the more moist its surface, the greater the loss through volatilisation. The quantitative significance of volatilisation is uncertain (THS 1983).

Run-off: Run-off is defined as the movement of water across the soil surface into water channels. Clays are most likely to lose nitrates as run-off, although the cumulative amount of run-off and leachate lost may be the same (Conway and Pretty 1991).

Leaching: Leaching is the process whereby water containing dissolved soil materials infiltrates the soil profile. Leaching losses of N occur when soils have more incoming water (rain or irrigation) than the soil can hold. As water moves through the soil, the nitrate NO_3^- in soil solution moves along with the water. Ammonium NH_4^+ forms of N have a positive charge and are held by the negative sites on the clay in the soil; therefore, NH_4^+ forms of N leach very little. In sands where there is very little clay, ammonium forms of N can leach. Coarse-textured sands and some 'muck soils' are the only soils where ammonium leaching may be significant. Overall leaching is comprised of mainly of NO_3^- and is greatest in light soils and soils with coarse structure i.e. higher infiltration. Leaching occurs throughout the year although in the UK leaching rates are greatest in winter when soil moisture is at field capacity and precipitation exceeds evapotranspiration (THS 1983). The leached nitrogen in the soil may either find its way down into an underground aquifer or any period of time (ranging from months to years depending on catchment specific geology) or like run-off make its way through field drainage ditches to surface water.

Nitrogen leaching depends primarily on the weather, i.e. precipitation and evaporation, and varies for every crop/soil combination. The weather determines the crop's ability to utilise the available nitrogen for growth and hence also determines the amount left over for leaching. Of this 'potential leachate' the actual leachate and run-off depends on the water movement down and across through the soil respectively. This is turn varies for every crop/soil combination and is primarily a function of precipitation, potential evaporation and physical features such as slope etc. It is no surprise that the leaching pattern follows the seasonal distribution of rainfall, with the greatest losses occurring in late autumn and over winter where there is little plant uptake (i.e. growth) and plentiful rainfall. Leaching and run-off are both affected by the type, frequency, timing and method of fertiliser application.

The direct contribution of fertilisers to nitrate leaching is surprisingly a controversial issue in the literature (even if not evident in the formulation of policies). There is definitely a positive correlation between the increased consumption of fertiliser and nitrate leaching (Davies and Sylvester-Bradley 1995). It is argued that N fertilisers increase soil organic matter which is subsequently broken down by microbes resulting in nitrate leaching (Addiscott et al. 1991).

On average only about 2% of N fertiliser is left in the soil as nitrates after plant assimilation and that the nitrate leaching in autumn is the consequence of mineralised soil organic matter (Burt and Haycock 1993). In autumn the soil moisture content and temperatures favour microbial activity and the breaking down of crop and root residue or humus. Thus even the most 'judicious' use of nitrogen fertilisers contribute indirectly to nitrate leaching (Johnston 1994).

Leaching to groundwater is a complicated dynamic process which may take anything from a couple of months to even centuries depending on the physical and chemical properties of catchment geology (i.e. intervening rock layers). In fact elevated nitrate levels in boreholes in the South of England are attributed to the ploughing up of permanent pastures during the Second World War for cereal production (Hanley 1991).

2.5 Health Effects

The discovery of nitrates and pesticides in ground water during the 1970s dispelled the commonly held view that ground water was protected from agro-chemicals by layers of rock, soil and clay. This initiated what would become intense public interest in environmental protection and agricultural pollution because not only is water essential but its purity is a gauge of the quality of life (Braden and Segerson 1993).

A USA Environmental Protection Agency publication (EPA 1991) states "Only two substances for which standards have been set pose an immediate threat to health whenever they are exceeded: bacteria and *nitrate*."

Nitrate is a normal component of the human diet, with an average daily intake from all sources approximated at 75 mg. About 5% of the nitrate ingested by adults is converted (reduced) to nitrite by bacteria in saliva; this conversion continues inside the alimentary tract. A high pH (non-acidic) of gastric fluids favours the growth of nitrate-reducing bacteria. Nitrites in the stomach can react with food proteins to form N-nitroso compounds, which are carcinogenic in rats. The evidence for humans is not conclusive as various world-wide epidemiological studies have produced conflicting results (Croll and Hayes 1988). Nitrate concentrations in UK drinking waters do not pose a significant risk of stomach cancer (O'Riordan and Bentham 1993). However it is interesting to note that in the United States, where the drinking water nitrate standard is considerably more stringent than its EU counterpart (US 10 mg/litre; EU 50 mg/litre), the environmental research authorities have suggested nitrates in foods and water as possible sources of cancer in recent reports (NRC 1995; Environmental-Working-Group 1996)[2].

Human babies are extremely susceptible to acute nitrate poisoning because they have more nitrite forming bacteria in their digestive system. Nitrite reacts with haemoglobin (the oxygen carrying iron-containing blood protein) to form methemoglobin (incapable of transporting oxygen and naturally founds at low levels 0.5 - 2% of blood). The most obvious symptom of nitrate poisoning is a bluish coloration of the skin due to lack of oxygen in the blood, a symptom termed *cyanosis* (10% methemoglobin in blood). As the oxygen level in the blood diminishes, the baby is suffocated, this condition occurs at 50 -60 % methemoglobin level in their blood and is termed *methemoglobinemia* or 'blue baby syndrome'. Infants under the age of 6 months are most susceptible as they have relatively non-acidic stomachs which favour bacterial growth. The last death from methemoglobinemia in the UK was in 1950 and last reported case in 1972 (Addiscott et al. 1991). Although 3,000 cases have been reported world wide since 1945 the condition is believed non-existent in Western Europe (House-of-Lords 1989a).

On a cautionary note US Environmental working group report concludes that the incidence of methemoglobinemia is under reported and the potential link between nitrate exposure and stomach cancer needs to be reconsidered (Cook et al. 1996). The same report potentially links the long term impact of nitrate ingestion with a variety of conditions such as birth defects, thyroid hypertrophy, hypertension, and 15 kinds of cancer (including non-Hodgkin lymphoma (NCI 1996)). Very recently, research by Glasgow University has linked gullet cancer with nitrates in fruit and vegetables (McKie 2002). Gullet cancer, which is more common than stomach cancer, kills more than 3,000 people in the UK every year and there has been a threefold increase over the past 20 years.

2.6 Environmental Effects

Eutrophication is the enrichment of water by nutrients (such as nitrate or phosphate) which results in a disturbance in the balance of organisms due to an accelerated growth of algae and higher forms of plant life. When plants and algae die their remains sink and are consumed by *aerobic* bacteria. This results in a reduction of the level of dissolved oxygen. Eventually, often near the bottom of a lake, virtually no oxygen remains (*anoxic*) and aerobic species die. Under these conditions *anaerobic* bacteria flourish. Anaerobic bacteria often produce toxic foul-smelling compounds such as hydrogen sulphide (H_2S), thioalcolohs (RSH) and ammonia (NH_3).

Eutrophication is a natural process that occurs to all lakes over time as the weathering of rocks and soils from the surrounding catchment area leads to an accumulation of nutrients in the water and associated sediments. Young lakes (and man made reservoirs) usually have low levels of nutrients and correspondingly low levels of biological activity (*oligotropic*). Old lakes usually have high levels of nutrients and correspondingly high levels of biological activity (*eutrophic*). The

[2] Interestingly, the intensity of fertiliser consumption in the EC is illustrated by the fact that North America which has a much larger land base than the EC consumes less fertiliser (Liapis 1994)

natural time scale from oligotrophic to eutrophic is of the order of thousands of years. However, a high rate of input of nutrients (from human activities) can increase the rate of aging significantly resulting in eutrophic conditions developing after only a few decades.

Algal growth depends on temperature, convectional currents and nutrient supply - which are often the limiting factor (House-of-Lords 1989a). However these nutrients are provided by human activities such as run-off and leaching of inorganic fertiliser and manure (containing nitrates, phosphates and ammonia), run-off from erosion (following mining, construction work or poor land use) discharge of detergents (containing phosphates) discharge of partially treated or untreated sewage (containing nitrates and phosphates). Although both phosphorus and nitrogen are essential to algal growth, normally the former is the 'limiting nutrient' in freshwaters and the latter in marine waters (House-of-Lords 1989a).

A eutrophic aquatic ecosystem may (DEFRA 2002b):

1. Become unsuitable habitat for fish and invertebrates with reduced species diversity of both the aquatic habitat and other species higher in the food chain. Normally there is a change in dominant biota (e.g. carp replace trout and blue-green algae replace normal algae) and an overall increase in plant and animal biomass.
2. Become too low in oxygen (anoxic conditions) for species such as fish and shellfish.
3. Damage the quality of areas of high wildlife conservation value, such as Sites of Special Scientific Interest (SSSIs).
4. Increase the growth of rooted plants (e.g. reeds), water turbidity, sedimentation and frequency of algal blooms. Toxic algal blooms (e.g. benthic algae) which poison fish and shellfish, making them unsafe for people to eat and damage the fishing industry (however there is no well established link between nutrient enrichment and the incidences of shellfish toxicity in marine waters). Local livestock and wildlife may be at risk and

blooms in recreational waters can result in closure of the areas, with impacts on tourism.

5. Produce so much vegetation that navigation or recreational use of waters becomes impossible.

Applying excessively high levels of nitrates has been related to changes in the species composition of grasslands (Skinner et al. 1997). This is because grass species differ in their nutrient requirement and their ability to uptake ions. The Rothamsted Experimental Station investigated the impact of applying high levels of ammonium sulphate and found it to acidify the soil resulting in grass dominated low-diversity swards (Smith 1987). Nitrate fertiliser is also responsible for the dominance of sweet-grass reed across grazed wetlands in Cambridge and the weakening of reed stems in the Norfolk Broads (House-of-Lords 1989b).

Regarding the impact of nitrates on animal husbandry, high nitrate concentrations in water and feed lead to reduced vitality and increased stillbirth, low birth weight, and slow weight gain in livestock (NRC 1972). One estimate of losses to Norwegian Salmon and Trout farmers from eutrophication in the Baltic Sea is $200 million (Saull 1990). There are numerous reviews of CVM cost benefit analyses which attempt to calculate the external cost of nitrate pollution in rivers (Hanley, N and Spash, C. 1993).

2.7 Diffuse Pollution

Diffuse or Nonpoint source pollution is that without a single point of origin or not introduced into a receiving body (e.g. stream) from a specific outlet or discrete source (EEA 2002). Diffuse agricultural pollution is Scotland's second most significant cause of ground and surface water pollution after sewage. By 2010 it will be the most significant cause and admittedly the most difficult to address (Henton March 2000). The total 'external' environmental cost of agriculture was estimated at £2.3 billion in 1996 across the UK (Pretty 2000). Significant costs were attributed to contamination of drinking water with pesticides (£120 m/year), nitrate (£16 m), *Cryptosporidium* (£23 m) and phosphate and soil (£55 m).

Other sources of water pollution from agriculture include pathogens draining into water courses (which impacts on bathing waters), soil erosion (which affects water habitats and quality) farm tips, on-farm burial of carcasses and emissions of ammonia from intensive livestock units. Downstream agricultural processing plants such as soil seed crushing or vegetable washing plant have the potential to cause pollution as well (Henton March 2000).

2.8 Policy Response

The problem of diffuse or 'nonpoint source' nitrogen pollution is complicated because it a) overlaps different disciplines b) is influenced by stochastic natural processes, and c) nitrate policy has significant socio-political consequences i.e. numerous conflicting and influential interest groups promote their perspective. In short it involves complex interactions across bio-physical economic and social systems (Watson et al. 1996) which makes constructing a policy all the more difficult.

The problem of diffuse pollution from nitrogen has been widely recognised and partially addressed in the Scotland (SEPA 1999; Darcy et al. 2000). The United Kingdom's diffuse nitrogen pollution control policy has been driven primarily by European Union requirements. The following is an outline of various schemes, policies and directives in response to nitrate pollution.

2.8.1 Nitrate Sensitive Areas

A Nitrate Sensitive Area (NSA) is an area where nitrate concentrations in sources of public drinking water exceed, or at the risk of exceeding, the limit of 50mg/l laid down under the 1980 EC (European Community) Drinking Water Directive (80/778/EEC) – which came into effect in 1985 in the UK. The NSA scheme aimed to reduce or stabilise high and/or rising nitrate levels through *voluntary compensated* agricultural measures going beyond good agricultural practice (DEFRA 1999b). The NSA scheme involved whole or part-fields being entered for a five year period. It was available in 32 areas of England (10 pilot areas introduced in 1990 and 22 further areas in 1994) and covered 35,000 ha of eligible land. The scheme attracted 25,000 ha of land (71 %) of the designated area. An NSA scheme was not introduced in Scotland. Some details of the NSA scheme options available to farmers are presented in table 2.4.

The scheme was monitored by collecting data on cropping and husbandry practices for each field and using a computer model (NITCAT and MANNER) to estimate N leaching. This was accompanied by actual leaching measurement of representative fields (soil water sampling) and borehole concentrations.

225 farmers in the eligible scheme area were interviewed (ENTEC 1988) in a comprehensive study commissioned by MAFF. The report concludes that the scheme has 'contributed significantly' to nitrate reduction, however:

'Around *44% of participants sampled were unaware of their farm practices which contributed most to nitrate leaching. The majority of those who were aware stated they would not continue with the changes that the NSA scheme required if they were no longer constrained...the scheme has not altered farmer's attitudes to the extent that would make a compensated scheme unnecessary*'(ENTEC 1988).

2.9.2 Nitrate Vulnerable Zones

The NSA measures were quite separate from the *mandatory and uncompensated* measures introduced under the EC Nitrates Directive (91/676/EEC), which came into force in Dec 1998. The NVZ measures, which restrict the quantity and timing of applications of nitrogen fertilisers and livestock manures, equate to Good Agricultural Practice and are therefore uncompensated (DEFRA 1999b).

There are currently two NVZs in Scotland. The first was designated at Balmalcolm in Fife during 1996 initial directive implementation, while the second, the Ythan Estuary in North East Scotland, was declared an NVZ by the Scottish Executive on advice from the Scottish Environmental Protection Agency (SEPA) in 2000. SEPA believes high levels of macrophyte growth and changes to fauna in the Ythan estuary are attributable to excessively high nitrate concentrations (Scottish-Executive 1999).

The Nitrate Directive requires that an Action Programme of appropriate measures should be drawn up and made compulsory in the NVZs. Member States retain some

Table 2.4: NSA Scheme Options

There are a total of nine options available within the NSA scheme under three main classifications as follows (ENTEC 1988):

1) Premium Arable Scheme – Involves the conversion of arable land to extensive grass under one of the following management system:

Option A
Unfertilised, ungrazed grassland

Option B
Unfertilised, ungrazed grassland with species-rich seed mixture

Option C
Unfertilised grassland with optional grazing

Option D
Grassland with optional grazing and the application of up to 150kg/ha of nitrogen fertiliser per year (kg of nutrient)

Option E
Grassland with woodland

Setaside Option
Unfertilised, ungrazed grassland that can count towards meeting setaside obligations under the Arable Area Payment Scheme (AAPS)

2) Premium Grass - Involves the extensification of existing intensively managed grassland. To be eligible the grassland must have been receiving more than 250kg/ha per year of inorganic nitrogen in each of the previous three years.

3) Basic Scheme – allows a continuation under conditions designed to reduce nitrate leaching.

Option A
Low nitrogen Arable Cropping – Restricted rotation, i.e. potatoes or vegetable brassica crops not to be grown. Nitrogen is limited to the lower of either the economic optimum or 150 kg/N/ha in an one year

Option B
Low nitrogen Arable Cropping – Standard Notation. Nitrogen is limited to four years out of the five to the lower of the economic optimum or 150 kg N/ha in any year. In one year out of the five the Nitrogen limited is raised to 200kg N/ha.

discretion over the content of Action Programmes, but, as required by the Directive, there are certain types of agricultural controls that all Action Programmes must contain. In order of impact, the main points are as follows:

1) Farmers in the NVZs will be required to limit their applications of manure. The limit will be set initially at 210kg of total nitrogen per hectare and later reduced to 170kg.

2) Farmers will have to ensure that they have adequate manure storage capacity to allow them to observe closed periods for the application of manure.

3) Farmers will have to limit their applications of inorganic fertilisers to levels which are consistent with the nitrogen requirement of the crop, after allowance for nitrogen from residues in the soil and from other sources.

4) Farmers will be required to keep fertiliser and manure records, including the timing and level of applications.

The Scottish Executive has initiated the designation of further NVZs (in Aberdeenshire, Banff and Buchan, parts of Strathmore, Fife, Mid and East Lothian and parts of the Borders) by issuing a public consultation paper (Scottish-Executive 2002). The consultation states the proposed designation of approximately 18% of Scotland as NVZs. This is based on research by the British Geological Survey commissioned report (Ball and MacDonald 2001) which determines the groundwater nitrate vulnerable zones for Scotland utilising an *estimation of the risk of nitrate leaching* from land classification (Lilly et al. 2001). The report also categorised surface water catchments for the vulnerable zones and validated the risk methodology by comparing the results with actual concentration measurements by SEPA, Water Authorities data, Private Water Supplies data and miscellaneous datasets.

Furthermore the above mentioned consultation paper states the Scottish Executive will increase the designated areas (by including the Black Isle, Coastal Easter Ross, Nairn, Moray, parts of Strathmore, Falkirk and West Lothian) if actual data (currently being gathered) can support the risk analysis. Clearly non-point nitrogen pollution is a major and topical concern in Scotland.

2.10 Common Agricultural Policy (CAP)

The Nitrate Directive can also be viewed as a result of the greening process of CAP, initiated by the MacSharry reform of 1992 and reinforced by Agenda 2000. Greening can be defined as the process by which modern environmental symbols have become prominent in social discourse and policy rhetoric. Whereas environmentalism is the greening of institutions and their practices in terms of incorporating environmental considerations in political and economic decisions, educational and scientific research institutions and geopolitics (Buttel 1993). In the nineties there was a lot of work on the greening of agricultural policy (Winter 1996; Clark et al. 1997; Robinson 1997).

The Treaty of Rome established the foundations of the Common Market in 1957 which were later developed in the Stresa conference (1958). In the six member states at the time there was considerable state intervention, particularly regarding what was produced, price setting, marketing products and farm structures. If agricultural produce was to be included in the free movement of goods while maintaining State intervention in the agriculture sector, national intervention mechanisms incompatible with free mobility had to be either removed or transferred to Community level. The CAP was successful in the sense that the Community was soon able to overcome the food shortages of the 1950s, achieved self-sufficiency and later generating cyclical and structural surpluses. This was achieved by alignment of prices in the community above the world price through a system of guaranteed price support (e.g. intervention buying), import taxation, export rebates, community preference measures etc. It must be noted that there was no explicit reference to environmental considerations. It was generally thought that farming practices did not damage the countryside and agriculture was the mainstay of rural life. In fact a naively romantic and idealised view of farming pervaded (Lowe et al. 1997).

2.11 CAP Reforms

A recent DEFRA commissioned report investigating the environmental effects of CAP (JNCC 2002) has attributed the following agricultural changes to CAP:

1. Intensification: High guaranteed prices encouraged farmers to raise yields by increasing their use of fertilisers, pesticides and higher stocking densities. This has led to a decrease in the area of semi-natural habitats, reduction in the populations of associated wildlife species, and less landscape diversity. The amount of utilised agricultural land has been increased by removing hedges, walls, farm ponds etc. Similarly the use of larger machinery has damaged soil structure and functionality.

2. Marginalisation: Land of poor agricultural quality has traditionally been under mixed and low-productivity livestock systems. However under the CAP the poor financial reward to such enterprises has resulted in either intensification or the pursuit of some other economic activity. Farmers even resorted to moorland, heatherland, and wetland reclamation with the eventual loss of semi-natural vegetation and its biodiversity.

3. Specialisation: Through market interventions such as subsidies and quotas the CAP has encouraged specialisation of livestock enterprises (dairy farming) and particular crops (e.g. cereals, oilseeds and peas/beans). This encouraged monocultures and the phasing out of mixed farming systems which have negatively impacted biodiversity and landscape character.

4. Abandonment: Land with poor infrastructure, low economic productivity, declining populations, and low agricultural productivity has seen the abandonment of farmed land. Although this is more prevalent in southern member states and France, parts of the UK the switch from farming to forestry is an example of land abandonment. It must be noted that this may actually favour the environment, as forestry confers some wildlife benefits and generates less nonpoint source pollution than cropping.

Additionally since agricultural support is reflected in land value, farmers face rising land prices and rents. Thus increasing the opportunity cost of uncultivated and unimproved land and providing incentives for further reclamation, drainage, and intensification (Potter 1998).

Realisation of the above along with the fact that CAP was responsible for: a) rising budgetary costs; b) build up and storage cost of vast food surplus stocks; c) farmers benefiting at the expense tax payers and consumers; d) price support is an inefficient way of supporting farm incomes; e) declining farm incomes; and f) CAP's severe criticism for lowering world prices at GATT negotiations and the effect on the EC's external trade relations led to the *CAP reform settlement* (a watered down version of the MacSharry proposals) in 1992.

Support prices of cereals, oilseed rape and protein crops were gradually reduced by 29%, and in compensation farmers received direct payments per hectare sown under the Arable Area Payment Scheme (AAPS) provided a certain small proportion of land is setaside. Payment rates vary for different crops and reflect historic yield. In 1996/97 payments ranged from £236 per ha (for cereals in the Less Favourable Areas (LFA)) to about £480 per ha (for oilseeds on non-LFA land). Expenditure under the AAPS in Scotland in 1997 was around £143 million (Scottish-Executive 1998).

Similarly the price of beef was gradually reduced by 15% and compensation in the form of direct aid to producers was offered in the form of the Beef Special Premium Scheme (BSPS) and the Suckler Cow Premium Scheme (SCPS) based on stocking density. There are some limits on the extensification premium in the form of individual producer quotas and Scottish regional ceilings. Scottish expenditure in 1997 for BSPS was £28.1 million and £58.1 for SCPS (Scottish-Executive 1998). A Sheep Annual Premium (SAP) was also introduced. It should be noted that figures for 1997/98 have been quoted because the empirical modelling for this research was calibrated to the market prices in this period.

More interestingly, *Agri-Environmental* regulations (EEC No: 2078/92) were agreed upon as an accompanying measure to the 1992 reforms to encourage environmentally friendly farming. The Scottish programme incorporated the *existing Environmentally Sensitive Area (ESA) scheme* (see below) and the new Habitats, Heather Moorland, Organic aid and the Set-Aside Access schemes. Given the above pressures to reform CAP it must be remembered that the emergence of agri-environmental policy should not be attributed to purely environmental considerations (Baldock and Lowe 1996).

2.12 Environmentally Sensitive Areas

Environmental considerations and the emerging issue of sustainability had earlier resulted in the first Environmental Action Programme (EAP) in 1972 which stated that economic growth was *not an aim in itself.* Henceforth, environmental provisions were included in subsequent regulations. Article 19 of EC regulation 797/85 laid the foundations of Environmentally Sensitive Area (ESA) scheme (introduced in 1986) considered the first European agri-environmental policy measure (Lowe and Whitby 1997). Within an ESA, farmers and crofters are eligible to receive support payments which encourage traditional land management practices, and also to care for and enhance features of national and cultural heritage interest. There is provision for improvement for public access to enjoy the countryside. Participation in the scheme is voluntary and for a fixed time period. Payments to farmers depend on the forgone income and hence vary considerably, reflecting agricultural conditions and the particulars of the agreement drawn up. ESA agreements are flexible and offer higher payment rates per hectare provided farmers accept greater restrictions or agree to more stringent environmental considerations. As farmers are induced to engage in proactive environmental conservation (Crabtree and Chalmers 1994) ESAs actually recognise farming can provide environmental goods besides food.

Ten areas of Scotland, where traditional agricultural practices have been an important factor in maintaining land for its particular conservation, landscape, natural or cultural heritage interests, have been classed ESAs. These include very extensive areas, for example the two ESAs in the Southern Uplands together cover over half a million hectares. Across Scotland they measure in total 1.4 million hectares or 19% of agricultural land (Scottish-Executive 1998). In comparison there are 22 ESA designated areas in England covering 1.2 million hectares of land (EU 2002). A contingent valuation (CVM) and stated preference (SP) study attempted to quantify in monetary terms the conservation benefits of two ESAs in Scotland and concluded 'for both of the ESAs the money value of benefits is large.... the most conservative estimate of benefits outweigh the costs' (Wynn 1999).

2.13 Agenda 2000 CAP Reforms

Given the enlargement of the EU (hence the prospect of growing surpluses) and the pressures of the next round of the World Trade Organisation (formerly GATT) further reform of CAP under the Agenda 2000 has been imposed. These include further reductions in beef, dairy and cereal support prices with an increase in direct compensation payments. Compulsory setaside would be abolished, however compensated voluntary setaside would continue and '*member states would have flexibility to subject area aid for crops and setaside to environmental considerations'* (Scottish-Executive 1998). There is notably greater emphasis on targeted agri-environmental considerations (even in Rural Development Regulation (RDR)) and especially *horizontal measures* (Article 3 of Council Regulation (EC) No 1259/1999) such as cross-compliance (Scottish-Parliment 1999). In effect it stipulates that *all future regimes* and direct aid schemes under the first pillar of CAP must be developed in compliance with environmental standards. There have been numerous public consultations on the proposed reforms (DEFRA 1999a). In response there are proposals for a Scotland Agri-environmental Scheme (to replace the current ESA and Country Side Premium Scheme) which will offer payments to farmers who choose to practice conservation (Scottish-Parliment 1999).

Although the 1992 and Agenda 2000 CAP reforms have partially redressed the environmental impact of agriculture, little progress has been made in correcting *past damage* from artificially high prices for cereals sustained by the CAP (JNCC 2002). Similarly, even though the 1992 reforms and the introduction of the Arable Area Payment Scheme (AAPS) moderated the rate of loss of unimproved grassland to cereal production, the AAPS has effectively frozen the arable area and discourages reverting even marginal arable land to grassland through agri-environmental schemes (JNCC 2002).

2.14 EU Water Policy - Water Framework Directive

Concurrent with the evolution of Agricultural policy, there has been development of EU water policy. The first EC water directive was the Surface Water for Drinking Directive (75/440) which came into effect with the First Action Programme on the Environment. This was followed by the Surface Water Directive (75/464), the Groundwater Directive (80/68) and the Drinking Water Directive (80/778). The emphasis of these directives has shifted from the monitoring and regulation of point sources to nonpoint diffuse pollution (Ward 1998).

These disjointed directives have been revised and unified under the *Water Framework Directive* (WFD) 2000/60/EC (EU 2000). One purpose of the WFD is to give EU water policy a coherent and integrated approach. The European Environment Agency (EEA 1998) confirms that diffuse nutrient loads from agriculture are responsible for eutrophication of coastal water and nitrate

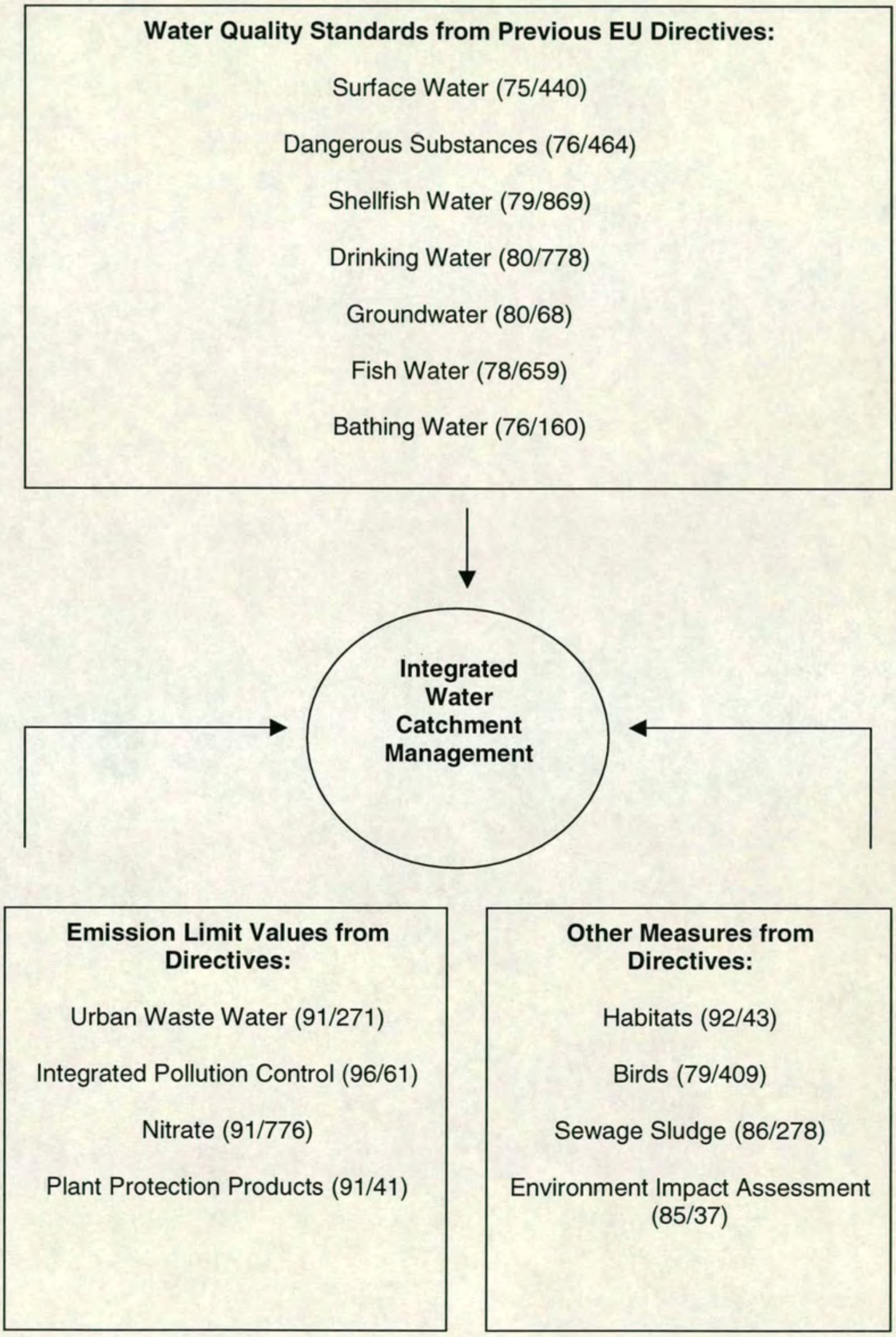
contamination of aquifers (EEA 1999); and indeed nitrate in E.U rivers has changed little since 1980. The WFD encompasses and supersedes the 11 existing EU water-related Directives[3]. In terms of policy implications the directive's two tenants require:

1. *Full Cost Recovery Pricing*. Member states are obliged under the WFD to implement the principle of 'full cost recovery pricing', as prioritised in the 5th Environmental Action Programme. This implies evaluating and accounting for '*environmental and resource costs*' (Article 9), i.e. agricultural *externalities* such as reduced river flows, diffuse nitrogen and pesticide pollution etc.

2. *Integrated Water Catchment Management*. This implies the organisation and regulation of water management at the 'catchment' or river basin level. River basins

[3] *Inter alia* the Bathing waters, Drinking Waters, Urban Waste Water, Nitrates and Habitat Directive.

Figure 2.2: Integration of Separate EU Directives into the WFD



comprise not only the surface run-off through streams and rivers to the sea, but the total area of land and sea together with associated ground and coastal waters.

The WFD, which must be transposed into national law no later than 2003, requires that all river basins and coastal waters must be assigned to a River Basin District (RBD[4]) and an overseeing competent authority designated. In Scotland it is proposed that this responsibility is bestowed on SEPA. Many stakeholders and respondents to public consultations have stressed the need for sub-basin (i.e. catchment level) planning as the key to integrated management (Scottish-Executive 2002)

While the UK is likely to be covered by approximately 14 River Basin Districts, each to become a single planning unit, much of the detail of catchment planning will be carried out at the Sub-Basin scale. Sub-Basins will be defined according to need, and so it can be reasonably expected that more detailed planning (smaller Sub-Basins or groups of Sub-Basins) will take place in areas of greater water stress. The East of Scotland is one such area, where rainfall and runoff are naturally low, and where the demand for irrigation water regularly exceeds availability, resulting in the use of artificial storages, dry stream problems and the creation of legislation targeted on managing these problems.

It is argued that delayed implementation would impose an administrative and scientific burden on the regulator between 2009 and 2012 only leaving three years by which in 2015 the environmental standards have to be met, thus creating '*significant uncertainty amongst those that might be the subject of regulation* ' (Scottish-Executive 2002). It concludes that in the interest of both business and regulators the Scottish Executive *should phase in the controls over an extended period from as early as 2005.*

2.15 Nitrate Politics

[4] The fundamental unit for applying the Directive's provisions

The complexity and disputes surrounding the Nitrate issue ranging from whether nitrogen pollution can be attributable to fertiliser use alone, uncertainty regarding the impact of nitrates on human and environmental health, the complexity of informational problems associated with nonpoint source (NPS) pollution etc., have all contributed to the delay and/or absence of effective nonpoint source policy formulation. Other cited explanations include a) agriculture was for many years more or less deliberately exonerated from stringent environmental regulation, and b) the relevance and widespread nature of NPS has only recently been fully comprehended (Dosi and Tomasi 1994a).

Until the late 1980's and early 1990s some still questioned whether the problem of diffuse nitrogen pollution was exaggerated and that the EU limit of 50 mg/litre was excessively low. Some authors at the time report fertiliser manufacturers denying the contribution of inorganic fertilisers to diffuse nitrogen pollution by arguing leaching is a highly variable natural process (Seymour et al. 1992). However currently the European Fertilisers Manufacturers Association (EFMA) website lists reasons why environmental taxation of nitrogen is inefficient and why the Code of Best Agricultural Practice, detailing nutrient budgets, fertiliser plans, rates, timing and type, is both "both environmentally and economically sustainable" (EFMA 1997). The National farmers union (NFU) has to publish research (questionnaires) on arable farming and their contribution to protecting the environment (NFU 2002), while also questioning the scientific validation of the procedure by which NVZs have been designated in Scotland (NFU-Scotland 2002) and proposing their own NVZ action plans.

There has been a gradual realisation of environmental concerns through education and the involvement of NGOs, Environmental pressure groups (such as WWF, Greenpeace etc.) and education through schooling. Likewise environmental groups, such as Greenpeace, which were previously against the use of economic incentives to control toxic discharges, now advocate the use of high taxation to phase out all nitrogen and pesticides (Clunies-Ross 1993).

In the late seventies and early eighties nitrate pollution was treated symptomatically i.e. it was considered a problem of the water industry and not of agriculture. Water treatment technologies and not prevention at the source was the conventional consensus. The change in attitude and policy came about because a) water industry privatisation led to a public debate and lobbying by environmental pressure groups which succeeded in the making the public aware of their exposure to the toxicity of nitrates (Maloney and Richardson 1994), b) change in EU agricultural policy in the aftermath of excessive surpluses and spiralling budgets (reasons for CAP reform) c) the gradual integration of environmental considerations in overall EU agricultural and water policy expressed as 'integrated water catchment management', d) sustained migration of 'environmentally concerned' people to the countryside has brought the issue of agricultural pollution to public attention (Ward et al. 1995).

Chapter 3
NPS Concepts, Controls and Caveats

3.1 Introduction

The purpose of this chapter is to explain the 'difficult' nature of diffuse agricultural pollution and its policy control measures, utilising both economic and non-economic instruments. There is considerable emphasis on the theoretical basis of economic controls and their efficiency. Some empirical works are mentioned in passing but the bulk of relevant empirical research is discussed in the next chapter. Non-economic approaches (regulatory standards and education), which do not provide a direct price incentive and are uniformly applied to all polluters irrespective of their contribution to pollution, are discussed at the end.

The fact that a great number of people contribute to agricultural pollution makes finding a practical and politically acceptable solution all the more difficult. Aggressive policies risk alienating important voters including 'family farmers' who have been traditionally venerated and supported as a matter of public policy, and whose historic land use rights have been accorded considerable deference (Braden and Lovejoy 1990). There is also the issue of public perception and recognition of agricultural NPS as a real and threatening environmental concern. Many agricultural NPS pollution problems are overshadowed by less pervasive environmental concerns, e.g. oil spills etc. (Braden and Lovejoy 1990). Some of the objectives sought through litigation are not always 'compelling', e.g. decreasing turbidity of streams and lakes, extending reservoir lives, improving diversity of aquatic biota, and enhancing the quality of recreational activities are not as 'dramatic' or health-threatening as accidental discharge of toxic chemicals, hazardous waste dumps, urban smog etc.

Public policies inevitably reconcile public expectation of 'good environment' with historic 'farming rights'. The irony is that increasingly the public does not want to pay for subsidies or inflated agricultural prices nor does it want to pick on 'family farmers' who are the alleged caretakers of their rustic ideal. Expectations about the proper balance are constantly changing, and formulated amid uncertainty concerning the value and cost of the policy outcome (Braden and Lovejoy 1990). The resistance to change is considerable. The conflict and difference of opinion over rights and environmental impact will have to be considerable before any meaningful change is undertaken.

3.2 Nonpoint Source Pollution

Agricultural nitrate pollution is an example of an environmental externality. An externality exists, '*whenever the welfare of some agent, either a firm or household, depends directly on his or her activities and on the activities under the control of some other agent as well*' (Tietenberg 1992). Externalities[5] can be both beneficial/positive (external economy) and detrimental/negative (external diseconomy).

Nonpoint source (NPS) or diffuse pollution is a stochastic and difficult process to understand. Thus before a theoretical discussion of control measures can be discussed its defined characteristics need to be explained:

a) Observability

Nitrogen water pollution is termed "nonpoint source" pollution because nitrate emissions (runoff, drainage, and leaching) from each farm (field or site) are diffuse. The problem is widespread and not confined to a few, easily identifiable producers. Runoff, by definition, does not originate from a single point, but leaves each site in so many places that accurate monitoring would be impractical or prohibitively expensive (Braden and Segerson 1993). Additionally pollutants running off one farmer's field may have originated from another farmer's fields further upslope, thus making it impossible to assign responsibility or enforce accountability (Braden and Lovejoy 1990).

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5 An unrelated externality termed *pecuniary*, arises when the external effect is transmitted through higher prices.

b) Variability

The inability to observe loadings would not be as troublesome if there were a strong *undisputed* correlation between ambient water quality and some observable aspect of production. However such correlations are unlikely, and where they can be established (e.g. input application procedure) they are unlikely to hold up across a range of conditions (Ribaudo et al. 1999). For example, an increase in river ambient nitrate concentration cannot be solely attributed to a farmer's inability to take appropriate measures because it may be a result of factors beyond his control e.g. high rainfall.

Thus the volume and concentration of runoff leaving a field depends not only on measurable factors, such as technology and inputs[6] used but also *stochastic* factors such as rainfall, wind, and temperature which are difficult to predict (Shortle and Abler 1997). Therefore ideally, policies should specify not only the ambient or emission target but also the frequency at which the standard (or goal) should be achieved (Shortle 1990b).

c) Spatial/Geographic Heterogeneity

NPS also varies by location due to differences in hydrologic characteristics, soil heterogeneity, land forms and other spatial characteristics. Efficiency requires accurate fate transport modelling i.e., site specific information. This is unlikely to be available or prohibitively complicated and/or expensive (Miltz et al. 1988)

d) Time lags

The time taken for NPS pollutants to move from the source to the receiving water body varies considerably from days (nitrate run-off to a nearby stream) to many decades (nitrate leaching to groundwater aquifers). This depends on local site

[6] Inputs are defined as those items used in production which can be applied in varying amounts (e.g. organic and inorganic fertiliser, pesticides, irrigation water etc.). Whereas, technology or management practices are defined as specific production methods e.g. conservation tillage, type and timing of fertiliser application, crop rotation, type of irrigation etc.

specific conditions and the properties of underlying rocks and geographical structures. This complicates policy formulation because the observed ambient levels which prompt regulation may be the result of past management practices or polluters no longer operating. Similarly, assessing the impact of remedial regulation may take decades before policy effectiveness can be determined.

d) Transboundary Effects

Many NPS pollutants, including nitrates, have long half lives and retain their toxicity far from where they originate. For example, nitrates entering the Gulf of Mexico via the Mississippi River have been traced to nitrate applications to corn crops in Minnesota, Iowa and Illinois (Goolsby and Battaglin 1993).

Given the above complexities the ability of the regulator entrusted with the daunting task of containing NPS pollution will invariably depend on how well the NPS process is comprehended.

3.3 NPS Process

The relationship between agricultural production and environmental impact on the receiving water body is complex and involves biological, physical and economic links. Efficient policy formulation depends on how well the following processes from input use to impact are understood and modelled:

1) *Pollutant generation*: The first link is the generation of nitrates which can potentially leach. This involves the nitrogen crop production function and the ability to relate the application of nitrogen to a particular crop/soil combination and predict the uptake, utilisation (plant growth) and soil nitrogen (N) transformations of biomass given the existing crop rotation. Notably the ability to accurately model other potentially limiting crop growth factors such as the availability and timing of water and sunlight are important in determining N uptake.

2) *Pollutant transport process*: The second link involves production practices and the movement of pollutants off a field. This is determined by rainfall, soil characteristics, distance to receiving water body, slope, chemical practices (method and timing of applications), irrigation type, crop management, agrichemical properties, conservation practices such as riparian buffers and constructed wetlands etc.

3) *Environmental impact*: This link involves the impact of discharged agricultural pollutants on the quality of receiving waters. Quality is expressed as bio-physical measures such as temperature, turbidity, pH, biochemical oxygen demand (BOD), ambient pollution concentration, fish populations, algae levels, zooplankton and bacterial concentrations, suspended particulates etc.

4) *Economic Valuation*: This relates how changes in water quality from ambient pollution translate into economic impacts (both use and nonuse). Changes in biological characteristic and physical appearance can have detrimental effects on anglers and people who use the water and its surrounding habitat for recreational use, including environmentalists who value the water body's ecology; similarly the treatment cost of municipal water use may rise. The value of service provided would, among other things, depend on overall demand by different users, regional population, income, and treatment costs.

Pollutant transport can take the form of a) *runoff*, which is the transport of pollutants over the soil surface by rainwater, melting snow or irrigation water that does not soak into the soil (nutrients may be *dissolved* (nitrate) or *adsorbed* to eroded soil particles (phosphate)), b) through deep *drainage ditches* along field boundaries, c) *run-in,* which is the movement of chemicals directly into groundwater though sinkholes, fractured or porous bedrock, or poorly constructed wells, or d) *leaching,* which is the movement of pollutants through the soil by any source of water.

3.4 Efficiency Considerations

First-best Solution

Ideally an *efficient* solution *maximises expected net economic benefits* – the net private benefits (NPB) of production minus the *expected* net economic cost of pollution assuming perfect information. NPB of production may include benefits to consumers and owners to factors of production (NPS policies may change input and output prices), but for simplicity here they only refer to aggregate profits. Decisions must also be made regarding damage *expectation* because it is impossible to predict the damages due to the natural variability in pollutant generation and transport, i.e. an *ex ante* solution as opposed to *actual or realised* outcome. The *first-best* optimal solution requires:

a) *Efficient input use*: the marginal net private benefit (MNPB) of using each input at every site must equal the expected marginal external damage from the use of that input. Thus the marginal input use will only result in a pareto-efficient outcome if it results in an equal increase in NPB and expected damage.

b) *Efficient land use*: A site (or field) should only be used for production if the profits exceed the expected social cost or external damage from pollution. *Marginal acreage* refers to sites with profits equal to (or smallest positive difference between) their expected contribution to damages *in the efficient solution.* Sites with a positive (negative) difference between profits and its expected damage are defined as *infra-marginal (extra-marginal).* Efficiency is only attained if infra-marginal and marginal land is under cultivation, extra-marginal land should be 'setaside' or take out of production i.e. forestry, grassland.

c) *Efficient technology use*: for each site technologies should be adopted such that the incremental impact of each technology (relative to the next best alternative) on expected net social benefits is greater than or equal to the incremental impact on expected damages.

The above three efficiency conditions imply that a socially acceptable level of water quality may require using fewer polluting inputs, retiring land or even abandoning conventional management practices.

Cost-Effective Solution

Damages from NPS pollution (or the external costs) are often largely unquantifiable, poorly understood and liable to legal challenges. Damage estimation would require valuation techniques such as travel cost or contingent valuation (Ribaudo and Hellerstein 1992; Crutchfield et al. 1997) but such exercises have been criticised as time consuming, costly and unreliable. Thus reducing expected damages may not be a measurable policy objective (Baumol and Oates 1988). Instead policy makers rely on reliable measurable proxies for expected damages including physical (ambient water concentration, expected runoff), and production related performance (input use or technology) indicators. These benchmarks act as socially desirable *standards*[7]; the notion of social optimality is replaced by cost-effectiveness or 'efficiency without optimality' (Baumol and Oates 1988). Due to the natural variability of NPS pollutants ambient and runoff targets/standards should be defined in terms of a likelihood of occurrence (Shortle 1990a).

Environmental policy is *cost-effective* if it achieves some measurable objective (standard) at least resource cost. However the efficiency conditions binding on first-best solutions are required under a cost-effective policy i.e. a) reduction (increase) in the use of inputs which increase (mitigate) emissions, b) adoption of appropriate technologies, c) and appropriate land use decisions at the extensive margin. Better goals are those which are more correlated with emissions and those which limit or have negligible pollution-increasing substitution effects.

Cost-effective policy solutions will generally differ from the *first-best solution* (Horan et al. 1998) as the only consideration is the cost of achieving the

[7] Policy objectives are specified in terms of a reduction in nitrogen fertiliser application rate (e.g. 30% reduction in catchment N use) or mandatory use of a particular tillage technology. Such specification exercises greater direct control because it is set deterministically and is easily verifiable.

predetermined environmental goal - benefits cannot be quantifiable. It is possible that a cost-effective policy meets the mean ambient water standard at least cost. However this method, while reducing the mean, may unintentionally increase the variability of emissions and increase damages (Shortle 1990a). Without the ability to measure damages it might not be possible to recognize when such situations arise.

Second-best Solution

Technically cost-effective policies are termed 'second best', however keeping with the convenient distinction used by some US department of Agriculture reports (Ribaudo et al. 1999) cost-effective policies after consideration of transaction costs can be termed *second-best*. Transaction costs vary for every instrument and include the costs of obtaining information, designing, administrating and enforcing a policy. Given these considerations a second-best policy is one which achieves the environmental standard at least cost.

The natural dichotomy of policies to control diffuse pollution comprises of *economic* and *non-economic controls*. Excluding political considerations, all policies should be judged on their 1) incentives, 2) relative complexity, 3) informational requirements (both principal/regulator and agent/farmer), 4) adaptability to changing economic and environmental conditions, and 5) operational transaction costs (i.e. administrative and enforcement costs) (Ribaudo et al. 1999). In this thesis the terms farm, firm, agent, polluter and site are used interchangeably unless stated otherwise.

3.5 Economic Incentives

Agricultural NPS pollution occurs at levels greater than the social optimum because of market failure, i.e. the market's inability to relay the social costs of pollution to producers. Economic incentive-based instruments create prices for the externalities by using taxes or subsidies (quantity rationing through marketable pollution permits (MPPs) effectively operates as a tax). Taxing expected emissions forces producers to 'buy' expected emissions from society and thus consider the social cost of pollution. The main benefits of economic incentive based policies are:

1) Producers are usually free to use whatever strategy is most profitable to their site specific conditions
2) Farm strategies can adapt to any change in relative prices of inputs and outputs or if new technologies become available
3) Abatement costs are likely to be lower with incentives than under command and control policies because producers can utilise site-specific knowledge to their advantage in reducing compliance costs. Regulators normally have limited farm specific information.
4) Economic controls provide incentives to develop (possibly market) new approaches to reducing pollution abatement costs.

Two distinct economic approaches to correcting externalities exist. The first *Pigouvian* approach (Pigou 1920) involves a regulator who attempts to correct the divergence between private and social costs of the externality. The second *Coasian* approach (Coase 1960) advocates free negotiation between affected parties, irrespective of the initial property right allocation, in order to *internalise* or resolve the externality. Unfortunately the Coasian approach is normally not an option when the transaction costs of negotiation are considered (Pearce and Turner 1990; Tietenberg 1992). Agricultural emissions in particular are normally characterised by numerous producers which render a Coasian bargaining solution infeasible[8]. This Ph.D. does not consider the Coasian approach.

Under ideal Pigouvian taxation, the authorities have complete information and can determine the *marginal social cost*, *marginal eternality cost*, and *marginal net private benefit* functions. With perfect information it is easy to levy a charge that brings private costs in line with social external cost; Pigou defined the optimal tax equal to the marginal external cost at the optimal level of pollution[9] (Pigou 1920). An intuitive explanation is obvious: for as long as the tax exceeds the marginal abatement cost, a cost minimising producer will opt for reduced emissions. An efficient solution would have the abatement costs at the margin equalised at all

[8] Detailed criticism of the Coasian approach is abundant in the literature (Varian 1993; Maki 1998).
[9] A diagrammatic explanation of pigouvian taxation can be found in most elementary environmental and natural resource economic textbooks, e.g. (Pearce and Turner 1990; Hanley et al. 1997).

sources, implying that high abatement cost sources reduce emissions by less than low abatement cost sources (Baumol and Oates 1988).

A mathematic treatment[10] of the first-best solution is presented. The formal proof of a tax's optimality does not require 'perfect competition', all it requires is that producers minimise the private costs of producing at which ever level of output it selects and has no monopsony power.

3.5.1 First-best Efficiency Conditions

Polluting emissions from the *i*th nonpoint source or 'land parcel/field/site' $(i = 1,...,n)$ are given by $r_i = r_i(x_i, \alpha_i, \upsilon_i, \theta_i, A_i)$, where r_i are nonpoint emissions, α_i represents site characteristics (e.g. soil type and topography), υ_i represents stochastic environmental variables (e.g. rainfall), θ_i represents parameters of pollution processes unique to site i. Discontinuous choices are represented by a scalar, A_i, which is referred to as the technology in use. The (m x 1) vector of inputs used on the *i*th land parcel is denoted by x_i.

Ambient pollution concentrations a are a function of non point emissions, watershed characteristics ψ, natural background pollutant levels ζ, transport parameters δ, and stochastic variables γ; therefore $a = a(r_1,...,r_n,\zeta,\psi,\delta,\gamma)$ and $\frac{\partial a}{\partial r_i} \geq 0$ for all i. All polluters are assumed to be *profit maximising, risk neutral and to have no influence on input or output prices*. The change in producer's quasi-rents (profits less fixed costs) is an appropriate measure of the costs of pollution control. The *i*th nonpoint source's expected profit function for any input choice *j* is $\pi_i(x_{ij})$. This is like a restricted profit function where the restrictions apply to inputs affecting nonpoint emissions. The social cost of pollution, *D*, is an increasing convex function of the

[10] In the main text and appendix of this chapter the mathematical notation is, unless stated otherwise, 'standard' and similar to that used extensively in the literature (Shortle and Dunn 1986; Shortle and Abler 1997; Shortle et al. 1998; Ribaudo et al. 1999; Shortle and Horan 2001)

ambient pollution concentration, i.e. $D = D(a)$, and $\frac{\partial D}{\partial a} > 0, \frac{\partial^2 D}{\partial^2 a} \geq 0$. It is also assumed that society is risk neutral.

The objective function restricted on technology is:

$$J(A) = \underset{x_{ij}, n}{Max} \sum_{i=1}^{n} \pi_1(x_{i,} A_i) - E\{D(a)\} \qquad \text{(EQ- 3.1)}$$

The necessary conditions for a maximum are:

Incremental impact of input usage:

$$\frac{\partial J}{\partial x_{ij}} = \frac{\partial \pi_i}{\partial x_{ij}} - E\left(\frac{\partial D}{\partial a}\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\right) = 0 \quad \forall i, j \qquad \text{(EQ-3.2)}$$

Incremental impact of nth site :

$$\frac{\Delta J}{\Delta n} \approx \pi_n(x_n, A_n) - E\{\Delta D(a)\} \approx 0 \qquad \text{(EQ-3.3)}$$

The external damage of employing extra land in cultivation must be considered in achieving the social optimum. Condition eq–3.2 equates marginal net private benefits from the use of input x_{ij} with expected marginal external damages from its use, thus ensuring pareto optimality. Thus the optimal pigouvian tax, t_{ij} would equal

$$t_{ij} = \frac{\partial \pi_i}{\partial x_{ij}} = E\left(\frac{\partial D}{\partial a}\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\right) \forall i, j. \qquad \text{(EQ-3.4)}$$

In eq-3.3, $\Delta D(a) = D(a(r_i,...,r_n,W)) - D(a(r_i,...,r_{n-1},W))$, i.e., the difference in damages with and without site *n*. If the nth site is optimal then the addition of another site will have a negative incremental impact.

Finally the optimal technology vector, A^*, is determined by solving for an efficient allocation for each possible value of A and comparing expected net benefits. Technology A^* is more efficient than technology A' when $J(A^*) - J(A') \geq J(A')$.

The optimal technology satisfies:

$$J(A^*) - J(A') \geq 0 \forall A' \qquad \text{(EQ- 3.5)}$$

this reduces to:

$$\begin{aligned}\pi(x_i(A_i),(A_{i*}) - \pi_i(x_i(A_i^{'}),A_i^{'}) &\geq E\left\{D(a(r_1^*,...,r_i^*,...,r_n^*,W))\right\} \\ &- E\left\{D(a(r_i^*,...,r_{i-1}^*,r_i(x_i(A_i^{'}),A_i^{'},v_i),r_{i+1}^*,...r_n^*,W))\right\}\forall A_i^{'}\end{aligned}$$

(EQ-3.6)

where $r_i^* = r_i\left(x_i\left(A_i^*\right), A_i^*, \alpha_i, \upsilon_i, \theta_i\right)$. Technology choice will be inefficient if its impact on the externality is ignored.

The inability to attain a first-best solution and the adoption of exogenously determined environmental standards instead (second-best solution) has been presented earlier. To reiterate, this is because in reality estimating the marginal net private benefit[11] and especially marginal external cost is problematic and contentious, not withstanding the multitude of economic valuation techniques (Hanley and Spash 1993). For a discussion on the theoretical issues regarding pigouvian taxation e.g. the entry/exit condition, incentives provided, and the prices versus quantity debate see the endnote[i].

Economic instruments in practice are divided into *performance-based incentives* i.e. those based on the results of farmers actions (runoff, measured ambient levels or environmental damage) and *design-based incentives* i.e. those based on farmer's actions (inputs and technology use)[12]. Ideally bases closely correlated to water quality (runoff and ambient levels) are preferred to those which are indirectly related, such as output (Braden and Segerson 1993). However information and transaction costs along with moral hazard issues may force the regulator to target other bases[13].

[11] *Adverse selection* and *moral hazard* are prevalent and incentives to shirk on abatement. Adverse selection is the inability of regulators to identify producer costs and how they vary, while moral hazard exists when the authority cannot observe the actions of a producer.

[12] Another dichotomy is *ex ante* policies, i.e. those which target choices before the polluting activity occurs, and *ex post*, i.e. those which come into effect once damages have occurred (Segerson 1996).

3.6 Performance-Based Incentives

The most logical tax/subsidy bases to target are the outcomes of producer decisions i.e. run-off (loadings[14]) or ambient concentration. In theory a corrective tax/subsidy on run-off leaving a site is akin to a price control on a point source discharge. However runoff cannot be monitored at reasonable cost given the current technology – rendering it an infeasible option at present. This leaves incentives based on ambient levels.

3.6.1 Ambient Taxation

Research on ambient incentive schemes approach the NPS problem as a *moral hazard* or "environmental shirking" issue. Assuming firm's input decisions are costly to monitor, their relationship with ambient levels uncertain and emissions are unobservable implies there is considerable uncertainty about the polluter's efforts to make socially desirable decisions.

Given the inability to effectively monitor producer efforts in the case of nonpoint pollution, Segerson (Segerson 1988) proposed an innovative tax based on overall ambient pollution concentration. She combined penalties and rewards for exceeding a specific level of total ambient concentration in a river catchment. The charge comprises a per unit tax based on the deviation from an exogenously determined ambient standard and a lump sum penalty for not meeting the standard. Firm specific taxes (subsides) are charged (paid) when the ambient pollution levels rises above (falls below) an exogenously determined target. The tax portion can be written as $t_i a + k_i$ where t_i is the ambient tax rate for firm i and k_i denotes a firm-specific lump sum tax or subsidy (*the derivation of a cost-effective ambient tax is presented in appendix 3.1*). Assuming profit maximisation, firm i will choose its inputs to maximise the following after tax profits:

$$V_i = \pi_{Ni}(x_i) - E_i(t_i a) - k_i \qquad \text{(EQ - 3.7)}$$

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[13] Choosing a base for point sources is not an issue as end of pipe discharges are directly related to water quality and easily observable at relatively low costs (Baumol and Oates 1988).

[14] Loadings refer to the actual pollutant entering the stream, i.e. what remains of the runoff generated at the field/site by the time it enters the river.

Where E_i represents a firm's expectation over all uncertain and stochastic variables. This expectations operator also extends over the ambient impacts of all other firms' choices as each firm's tax depends on the *overall* ambient level (Cabe and Herriges 1992). In reality, E_i (one farmer's expectation) may differ from the next farmers E_l (where $i \neq l$) and also from the regulator's expectation, denoted by E. Assuming Nash conjectures, (i.e. firm believe their input decisions are independent of other firm's decision) then firm i's profit maximising conditions for input use (interior solution) are:

$$\frac{\partial \pi_{Ni}}{\partial x_{ij}} = E_i\left(t_i \frac{\partial a_i}{\partial x_{ij}} \right) \qquad \forall i, j \qquad \text{(EQ - 3.8)}$$

Implying each firm equates its marginal profits from the use of an additional input with the expected marginal impact the input will have on the firm's tax. The first-best tax is determined by equating eq-3.7 with eq-3.2:

$$t_i = \frac{E\left(\frac{\partial D^*}{\partial a} \frac{\partial a^*}{\partial x_{ij}} \right)}{E\left(\frac{\partial a^*}{\partial x_{ij}} \right)} \qquad \forall i \qquad \text{(EQ - 3.9)}$$

The advantage of this system is that it does not require continual monitoring of individual emissions, as the liability of each polluter is tied to aggregate emissions in the waterway - not his own contribution. The polluters are told of a particular ambient toxic concentration which they must satisfy. If they fail to keep levels below the target each polluter faces the full brunt of the damage. Thus if there are n polluters and the environmental damage is £x, the regulatory authority collects a total of £nx. In paying the full marginal damage of the total ambient pollution, there is no incentive to free-ride on other producer's actions. As more money is collected in taxes from polluters than the actual damages this is *not budget balancing*.

Note the above is only the first-best tax when $E_i = E \quad \forall i$ and $m = 1$, and is *state-independent* because it does not depend on the realised values of random variables; it depends on the expectation over all possible values. However, if $E_i = E \quad \forall i$ and

$m > 1$, it can be proved that a state independent tax cannot be first-best (Horan et al. 1998) for the same reason that estimated emission taxation cannot be first best.

Horan (Horan et al. 1998) identified two ex ante efficient taxes for nonpoint polluters under less restrictive conditions than Segerson. Firstly, a *linear* state-dependent tax rate of the form $t = \partial D^* / \partial a$ applied uniformly across polluters. This tax is conditional on the realisation of all random variables and hence has no expectation operator. As only the realised values of stochastic processes determine *t*, therefore t is a random variable when firms make their decisions. Thus they face $E(ta)$ as opposed to $tE(a)$. Their second tax is a nonlinear function of ambient levels or $E(D(a))$, i.e. each firm pays an amount equalling total damages. As with a linear tax, the nonlinear tax is state-dependent and applied uniformly across all firms (Shortle et al. 1998).

By themselves, ambient incentives do not provide incentives for optimal entry or exit and require additional lump sum[15] instruments. A lump sum incentive would take the form of a tax on producers who cultivate extra-marginal fields or a subsidy to producers who voluntarily retire extra-marginal acreage. Under the tax if they do not produce they are not subject to taxation. Production on land under infra-marginal or marginal land is not subject to lump-sum tax or subsidy – unless their decision to produce is influenced by the ambient tax. A lump-sum rebate can reduce their tax burden without compromising cost-effectiveness. Producers are free to change their decisions in the face of changing environmental or economic conditions and also utilise any private information to reduce their compliance costs.

Ambient-based incentives are *seemingly* advantageous because ambient concentrations (instrument base) are closely related to the external damage and because ambient levels can be monitored without having to observe the action of

[15] A *lump sum* instrument is a fixed tax or subsidy which is usually used to influence *discrete choices* or to determine the *distributional impact* of policies. For example a lump sum subsidy may be made contingent on the adoption of conservation tillage. Those lump sums which are not contingent on particular actions are not applied to a base and therefore do not influence marginal incentives.

each producers. The incentive base depends on group performance. However ambient-based incentives can only be designed to achieve a *cost-effective outcome under highly restrictive and unreal assumptions*, i.e. all producers are risk neutral and they all share the same expectation about the stochastic nonpoint process. Risk-averse producers will not like the onus of risk associated with predicting stochastic nonpoint processes (Horan et al. 1998). Second-best ambient-based incentives can be devised given risk aversion and heterogeneous expectations about the nonpoint process. Potentially high transaction costs may lead to uniform second-best taxation across all producers.

The information requirement on producers and their expectations ($E_i = E \quad \forall i$) is not realistic by any stretch of the imagination; catchment farmers must be able to evaluate how their actions and the actions of other farms interact and affect the incentive base, i.e. ambient nitrate level. Similarly the regulatory authority needs to know each producer's belief structure about the inherently stochastic nonpoint process if it wants to evaluate the impact of a tax. Thus if there is asymmetric information regarding the profit and environmental types of firms (i.e., $E_i \neq E$, and $E_i \neq E_l \quad \forall i \neq l$) then incentives designed on the assumption of identical expectation will not be efficient. Outlines of numerous ambient taxation problems given the requirement for site specific information on complex pollutant fate & transport systems, are present in the literature (Cabe and Herriges 1992). More importantly the polluter's prior belief about the transport system is instrumental in determining whether a firm believes its effluents will alter ambient concentrations and hence their compliance. That is to say if the producer doesn't believe his production will impact on ambient concentrations, his actions will not change in the presence of a tax.

Another problem is that current ambient conditions may be more a function of past decisions than present ones, e.g. nitrates can take years to move from fields to aquifers (Kim et al. 1993). Secondly, monitoring ambient levels can be highly costly and subject to considerable error especially in the case of groundwater. There are also numerous political and legal limitations as farms which do abate at a private cost may be penalised due to environmental shirking on the part of others. Conversely, individuals can *free ride* on the abatement efforts of others and even be rewarded although they took no such effort themselves (Xepapadeas 1999).

Given the above it has been suggested (Weersink et al. 1998) that ambient taxation may be best suited to small catchments in which agriculture is the only pollutant source, farms are relatively homogeneous, water quality is readily monitored, and there are short time lags between actions and water quality impacts.

Numerous versions of the ambient charge are found in the literature (Hanley et al. 1997). Some have suggested that regulators can develop charge adjustment procedures that achieve ambient standards at multiple receptors at minimum costs[16], by using a Walrasian *tatonnement*[17] process an equilibrium vector of ambient and emission charges that are both cost-effective and environmentally friendly can be found (Ermoliev et al. 1996). Even in the absence of pollution control costs data, the regulator can use observations of excess pollution to adjust ambient charges in a stepwise way by monitoring responses to various charge levels and with the help of a data bank on transfer matrixes, calculate actual concentrations and optimal ambient standards.

In the 'learning stage', polluters are made to register expected emissions corresponding to charges. In the next stage, the agency verifies actual emissions, reserving the right to penalise discrepancies between expected and actual emissions, and administer charges accordingly. A gross oversimplification in this analysis is the assumption that polluters do not give erroneous strategic information (responses[18]) to influence charges.

[16] Baumol & Oates proposed an iterative adjustment of emission charges in successive steps (directed by difference between actual and target emissions) would lead to the target level at minimal costs (Baumol and Oates 1988). Others such as Bohm & Russel believe the least cost claim is suspect (Bohm and Russell 1985).

[17] Where an auctioneer discovers the equilibrium price before agents act and thus 'false' emission adjustment at non-equilibrium prices can be avoided.

[18] Exaggerate their emission control in the learning phase in order to obtain lower fees in the implementation phase.

Whereas in the presence of typical ‘adverse selection’, a firm is shown to have an unbounded incentive to over-report marginal clean up costs, work by Bulckaen (Bulckaen 1997) has shown that this result needs to be revised if firms are to behave ‘consistently’, with its own reports i.e. if the regluator requires firms to pay charges based on reported marginal clean-up costs.

3.6.2 Liability Rules

Individuals damaged monetarily or otherwise by the activity of polluters may be given the right to sue for damages[19]. Liability rules are essentially performance based incentives which are imposed after the damage is realised (Shavell 1987). Liability rules hold polluters liable for the environmental damage they impose. Although they are imposed *ex post* they effectively serve as an *ex ante* incentive to farms/firms as they will likely weigh the benefits of generating pollution against the penalties they may expect as a result of their actions. Liability rules may be divided into *strict liability*, where polluters are held absolutely liable for any damages, and a *negligence rule* where they are only liable if they failed to act with ‘due care’(Segerson 1995).

With NPS where multiple polluters are exist by definition, the principle of *‘joint and several liability’* dictates that damages be divided according to the court’s choosing, unless there is some legal precedent. If one producer is held responsible for damages then he may later sue other responsible parties to share the burden (Miceli and Segerson 1991). If a victim(s) has no way of protecting himself (themselves) from the pollutant, and only the polluter influences damages then the situation is termed *unilateral care*.

This approach undoubtedly favours polluters. It relies on concerned individuals or organisations to prove ecological damages (possibly involving estimations of intangible uses) present scientific results, and face the financial costs of litigation, before any liability is awarded, if any.

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[19] Which if successful may serve as a precedent for future actions

Under strict liability rules producers are uncertain whether they will be held liable for environmental damages. This is because ambient pollution is determined by the collective actions of all producers in a catchment, and because there is considerable uncertainty regarding pollutant transport and the ability to assign responsibility. Consequently each producer will have site-specific expectations about stochastic weather processes, other uncertain environmental factors, and most importantly being sued and then found guilty.

In theory strict liability can attain efficient NPS pollution control. Each polluter expects to pay the total expected damages from pollution, plus or minus a lump sum component that distributes payments across polluters - so that total payments equal total damages. It requires that the rule be site-specific and so account for each producer's belief about the NPS process and the probability of being sued/proven guilty. This incentive mechanism still requires lump sum transfers to producers operating on extra marginal land to ensure optimal entry and exit.

If the victim(s) can somehow protect himself (themselves) then the situation is termed *bilateral care* (Segerson 1995), e.g. where victims are able to purchase a filtration system to protect themselves from groundwater contamination. Thus if victims can take precautions that producers cannot take for them (Wetzstein and Center 1992), then it is possible that victims may sub-optimally protect themselves if they believe they can sue and collect damages. It is possible to derive a modified strict libility rules based on victim precaution requirements (Wetzstein and Center 1992). The nature of NPS pollution (inability to identify polluters, levy responsibility or measure damages) makes the liklihood of a producer being sued and held liable very slim under a strict liability rule.

3.6.2.1 Negligence Rules

Under negligence based liability a producer is guilty if he/she failed to operate under 'standards of due care', where due care is either specified as performance based outcome (e.g. ambient levels) or producer's decisions (usage of polluting inputs and technology). A negligence rule is more appropriate to NPS pollution because it is not necessary to prove an individual producer's contribution to total damages. Polluters may be held collectively responsible and liable for damages if ambient levels exceeded the standard. Such a negligence rule does not correct optimal entry and exit through lump sum transfers, because it only applies to producers operating at the time the damages occurred. Additionally producing at suboptimal levels to avoid liability could result in extra-marginal land being cultivated, resulting in inefficient land allocation (Miceli and Segerson 1993).

Where a liability rule is based on producer decisions farms are held liable if suboptimal technology (relatively more polluting) is used, and if more polluting inputs and/or less pollution mitigating inputs are used than are optimal. This is fairer as only those farmers not using acceptable production practices would be held responsible. However there are considerable transaction costs (monitoring and administrative) in ensuring compliance.

In the US, both Federal and State regulators hold producers liable for damages only if they fail to use legal chemicals in accordance with the manufacturer's instructions and codes of practice. Such negligence rules are grounded in the belief that producers have a the 'right to farm' as long as they comply with acceptable practices (Ribaudo et al. 1999).

The informational requirements of liability rules are considerably prohibitive i.e. producers must have 'realistic' beliefs about their collective effects on ambient levels, and the profit functions of *all sites* (including other producer's decisions). Additionally the regulator needs to monitor the practices and input decisions of *all producers* on all sites and estimate the NPS generation and transport mechanism to identify 'optimal' practices which define 'due care'.

In addition actual litigation itself may be prohibitively expensive and put people off (Shavell 1987). Due to their high transaction costs, liability rules are best suited to the control of hazardous materials or for infrequent occurrences such as accidental toxic spills (Shortle and Abler 1997).

3.6.2.2 Non-compliance Fees

Non-compliance fees are another type of liability rules which provide incentives to follow some prescribed mandate or technological restriction by raising the cost of misbehaviour. Xepapadeas tried to circumvent the problem of moral hazard (identifying the excessively polluting culprit) and constructed a combination of fines and subsidies that would provide an incentive not to violate the prescribed standard (Xepapadeas 1991). His model is again drawn on work pioneered by Holmstrom and so is an improved modification of Segerson's ambient charges (Holmstrom 1982).

If ambient concentration of pollutants at designated receptor sites exceeds target standards, the regulator selects one pollutant at *random* and fines him. A portion of the fine minus the environmental damage from non-compliance, is redistributed among the other polluters. If properly designed it significantly raises the expected cost of shirking. It does not require monitoring of individual polluters, their pollution abatement efforts or costs, and so in this respect it is an improvement on taxes/subsidies. Like ambient charges it requires monitoring at receptor sites only; unlike such charges it's budget balancing and does not require that punitive fines exceed the damage from non-compliance.

Later work demonstrated the caveat that random penalties will only work if *all* producers are risk-averse (Herriges et al. 1994). The reason being that balanced budgets create a interdependence among producers - one producer's loss is another's gain. As fines are raised there are two contradicting incentives in play - because a producer's incentives will be motivated by his own expected penalty and on the expected penalty suffered by others, which is in effect, a benefit to him. Only risk averse producers will be more afraid of losing profits than about receiving profits. So for polluter decisions to match social objectives, the scheme requires risk averse producers who will magnify the expected fraction of marginal costs enough to offset the full marginal benefits of shirking. For an extended mathematical model and explanation refer to source (Xepapadeas 1991).

The political unattractiveness of random fines is apparent. A variation is the *environmental rank-order tournament*, where polluters are ranked ordinally on the basis of their abatement efforts and technologies (Govindasamy et al. 1994). Deviations from prescribed ambient levels are blamed on the lowest rank polluters. Thus the fine is not randomly allotted, and rather falls squarely on those most likely to have over polluted. Alternatively, regulators might reward the highest ranking producers if the ambient concentration was better than prescribed standards. Potential problems may arise if the regulator's rankings are biased due to incorrect modelling of fate-transport systems. Incentives may also exist for the higher ranked firms to pollute, knowing well that they will not be charged for misconduct and in fact receive a benefit from the redistribution of the fine levied on lower ranked polluters. A mathematical proof can be found at the original source, however some textbooks provide a simplified version (Hanley et al. 1997).

3.7 Design Based Incentives

Design-based incentives target variable input use and technology, both of which are further removed from the damages than aforementioned performance incentive bases. Design-based incentives are subdivided into *expected emission-based* and *input or technology-based incentives.*

3.7.1 Expected Runoff Based Incentives

Expected runoff may actually be estimated *ex ante* using biophysical-economic simulation models. These models attempt to predict farming production and pollution control decisions. Farming decision variables such as technology and input management need to be monitored to estimate runoff. The incentive-base (expected runoff) is *actually designed based as it depends explicitly on inputs and technology.* This requires that *farmers understand how their production decision effect expected runoff in the eyes of the regulator* – an educational burden on the agency. Heterogeneous expectations about runoff generation among farmers do not matter (unlike with ambient taxation) because tax incentives are based solely on the regulator's expectation.

Ideally a runoff estimate should provide producers with information about the impact of their choices on *expected damages* from pollution. As runoff is subject to natural variability choices made to achieve a particular *mean runoff* do not correspond to *a unique level* of expected damage (two separate input decisions may result in the same mean emission but with different variance, and hence different damage). Thus runoff based incentives based on *unbiased emission estimates* do not generally provide enough information to producers about the external cost of their decision (Ribaudo et al. 1999).

Initial research in NPS economics assumed perfect information regarding firm production, control costs and emission estimates from monitoring input use (Griffin and Bromley 1982). By assuming emission estimates are perfect substitutes for measure emissions, i.e. no forecasting error, and that emissions were *not stochastic* they were able to construct first-best taxes and standards. How the regulator derives a cost-effective solution based on a mean run-off target is derived in Appendix B.

Later work by Shortle and Dunn (Shortle and Dunn 1986) investigated solutions under the realistic assumptions that biophysical simulation models *do not eliminate* all the uncertainty regarding emission generation , i.e. the process is stochastic. They demonstrated that a first-best allocation could not be obtained using a tax on an unbiased emission estimator except under highly restrictive conditions.

Let t_i be a firm-specific tax applied to estimated emissions from the *i*th site *(how t_i is derived by the regulator is shown in appendix 3.3)*. To enable the polluters to compute their tax liability the regulator announces the relationship $E(r_i)$, where *E* is the agency's expectations operator. After tax profits for each site is given by $\pi_i(x_i) - t_i E(r_i)$ and the first order necessary condition for input use (for an interior solution) are:

$$\frac{\partial \pi_i}{\partial x_{ij}} - t_i E\left(\frac{\partial r_i}{\partial x_{ij}}\right) = 0 \quad \forall i, j \qquad \text{(EQ – 3.10)}$$

comparing eq-3.10 with eq-3.2 implies the following condition (after some manipulation) must hold to obtain a least cost solution (Shortle and Horan 2001):

$$t_i = \frac{E\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r_i}\frac{\partial r_i^*}{\partial x_{ij}}\right)}{E\left(\frac{\partial r_i^*}{\partial x_{ij}}\right)} = E\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r_i}\right) + \frac{\operatorname{cov}\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r_i}, \frac{\partial r_i^*}{\partial x_{ij}}\right)}{E\left(\frac{\partial r_i^*}{\partial x_{ij}}\right)} \quad \forall i, j \qquad \text{(EQ-3.11)}$$

A charge on estimated emissions *only* provides firms with an incentive to control mean emissions. However, choices which reduce $E(r_i)$ do not necessarily reduce $E(D)$ when D is nonlinear (Shortle and Dunn 1986; Shortle 1990a). Only when a single input influences emissions (i.e. m=1) or when the covariance between marginal damages and marginal emissions is zero[20] for each input for each firm (i.e. $D = \sum E(r_i)$) do the input decisions that cost-effectively reduce mean emissions also cost-effectively reduce expected damages (Horan et al. 1998).

A first-best outcome would necessarily have to account for other relevant moments (i.e. variance, covariance) of a firm's impacts on ambient levels other than the mean. In reality *political, legal or transaction costs* may prevent the regulator from implementing site-specific incentives. These transaction costs pertain to the informational requirement of designing site-specific taxes, i.e. trying to induce producers to truthfully report their private information. In practice farmers retain private information about land production practices, land productivity and other site-specific characteristics which affect runoff or economic returns. Producers are likely not to share this information as they fear it might be used against them in the design of environmental policy.

Hence the adoption of a single incentive rate applied uniformly to each site. A second-best charge on a nonpoint emissions proxy would maximise social expected net benefits subject to firm responses to the tax. For simplicity, assume identical firms[21], (subscript i dropped); the optimal uniform tax will be:

$$t_r^* = \frac{\sum_{j=1}^{m}\left[E\left(\frac{\partial D^*}{\partial a}\right)E\left(\frac{\partial a^*}{\partial r}\right)E\left(\frac{\partial r^*}{\partial x_j}\right)+E\left(\frac{\partial r^*}{\partial x_j}\right)\operatorname{cov}\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r}\right)+\operatorname{cov}\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r},\frac{\partial r^*}{\partial x_j}\right)\right]\kappa_j^*}{\sum_{j=1}^{m}E\left(\frac{\partial r^*}{\partial x_j}\right)\kappa_j^*}$$

(EQ-3.12)

where, $\kappa_j^* = \left(\frac{\frac{\partial x_j^*}{\partial t_r}}{\sum_{j=1}^{m}\frac{\partial x_j^*}{\partial t_r}}\right)$ and r_i^* and a^* are functions of e_k^* and x_{ij}^*, which are the solutions to the second-best problem described above. Interestingly, since κ_j should be interpreted as a weight (as $\sum_{j=1}^{m}\kappa_j = 1$), the numerator of eq-3.12 is the expected marginal social cost of input use averaged across all inputs; while the denominator is the expected marginal contribution of input use in emissions production again averaged across all inputs. Now as the second-best tax, t_r^* is uniform across all inputs it does not give farms incentives enough to adjust consumption of each input based on its marginal environmental risk effect. The environmental risk-effects are represented as the two covariance terms, which results in the following conclusions:

1) If damages are *convex in a,* the covariance term $\operatorname{cov}\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r}\right)$ has the same sign as $\frac{\partial \operatorname{var}(a^*)}{\partial r}$; implying that if increased loadings increase the variance of

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[20] $\operatorname{cov}\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r_i},\frac{\partial r_i^*}{\partial x_{ij}}\right) = 0$

ambient pollution and hence damages, then average risk is increased and t_r^* is larger.

2) If damages are *convex in loadings*, then the covariance term $\operatorname{cov}\left(\frac{\partial D^*}{\partial a}\frac{\partial a^*}{\partial r}, \frac{\partial r^*}{\partial x_j}\right)$ is of the same sign as $\frac{\partial \operatorname{var}(r^*)}{\partial x_j}$. Therefore, increased input use increases the variance of nonpoint emissions and hence damages (averaged across all inputs), thus risk is increased and so must the tax t_r^*, all other things being equal (Shortle and Horan 2001).

Overall administrative cost would be high because the regulator has to monitor input use and technology for all sites and because authorities have to provide farmers with run-off relationships for each site. The instrument is 'fairly flexible' (Ribaudo et al. 1999) as producers may utilise any private knowledge to reduce costs (input consumption). However farmers only have incentives to use managerial practices which can be modelled and are recognised to decrease loadings by the regulator. As in the case of ambient taxation, the implementation of a cost-effective tax schedule may result in a suboptimal entry and exit. This may be corrected by providing lump sum incentives to producers who produce on marginal and extra-marginal sites.

3.7.2 Input and Technology-based Incentives

There is considerable evidence that input-based incentives can bring about resource allocation changes. Agricultural firms respond to changes in the cost of inputs, increasing the use of those that become relatively cheap, while conserving on those that become relatively expensive (Shaumway 1995). In fact there is evidence that in the long run, input price responsiveness is even greater, as technologies are developed to further conserve the use of expensive inputs in favour of relatively cheaper ones (Haymi and Ruttan 1985).

Ideally the first-best input tax/subsidy must target all inputs and technology choices; thus the site-specific incentive rate for each polluting input should equal the expected social cost of a marginal increase in the use of an input (Shortle and Abler 1997).

Therefore inputs which decrease pollution (e.g. a nitrogen inhibitor), i.e. whose social cost of marginal input use is negative, should be subsidised. This result was derived by the seminal work of Griffin and Bromley, assuming perfect knowledge of production and polluting relationships, i.e. *non-stochastic* (Griffin and Bromley 1982; Stevens 1988).

Latter research demonstrated the viability of input tax/subsidy schedules even in the presence of *stochastic* and *imperfectly estimated emissions*. Suppose the input tax rate on input j of firm i is t_{ij} thus expected profits after tax are $\pi_i(x_i) - \sum_{j=1}^{m} t_{ij} x_{ij}$.

Assuming the farmers choose inputs to maximise after-tax profits, the optimal marginal input taxes are (after considerable manipulation (Shortle et al. 1998; Shortle and Horan 2001)):

$$t_{ij} = E\left(\frac{\partial D^*}{\partial a}\right) E\left(\frac{\partial a^*}{\partial r_i}\right) E\left(\frac{\partial r_i^*}{\partial x_{ij}}\right) + E\left(\frac{\partial D^*}{\partial a}\right) \mathrm{cov}\left(\frac{\partial a^*}{\partial r_i} \frac{\partial r_i^*}{\partial x_{ij}}\right) + \mathrm{cov}\left(\frac{\partial D^*}{\partial a}, \frac{\partial a^*}{\partial r_i} \frac{\partial r_i^*}{\partial x_{ij}}\right)$$

$$\forall i, j \quad \text{(EQ-3.13)}$$

In theory the optimal tax rate for input j for each site i (or firm/farm) equals expected marginal damages, times the expected marginal increase in ambient pollution levels from each firm's emissions, times the expected marginal increase in emissions for increased use of input j, plus "two covariance terms that act as risk premiums or rewards" (Shortle and Horan 2001). The risk terms signs depend on the properties of the emission function and can be interpreted as those in EQ 3.12. Such a tax will not however guarantee cost-effective entry/exit, i.e. optimal number of producers and sites, and extra lump sum incentives are required (Shortle et al. 1998). The design of such incentives can be found in appendix 3.3. Overall under a cost-effective input based tax/subsidy farmers will have an incentive to utilise their private knowledge to further reduce costs, and alter decisions as economic and environmental conditions change.

3.7.2.1 Technology

The use of per unit input based incentives will not necessarily create incentives necessary to induce producers to adopt the efficient technology. Essentially linear instruments only account for the marginal impacts of each producer's choices, whereas *production technology has a non-marginal impact on damages*. Thus if a suboptimal technology is used, then input use may also be suboptimal since all production decisions are interdependent. Lump sum incentives contingent on technology choices (tax on sub-optimal adoption, subsidy on optimal adoption) can produce optimal adoption. Alternatively a cost-sharing approach can be used to induce optimal technology adoption in the presence of adjustment costs. Most farmers are reluctant to adopt new technology due to the costs of obtaining information, management and capital constraints and risk perceptions[22] (Nowak 1987).

The United States Department of Agriculture (USDA) offers farmers incentives to adopt conservation practices under the Water Quality Incentive Program[23] (WQIP) and the Environmental Quality Incentive Program (EQIP). These include payments to encourage non-structural management practices, such as conservation tillage technology. The payments offset any private losses, increased risk of uncertain yield, and other short-term adoption constraints (Ribaudo et al. 1999). However research has questioned the effectiveness of past incentive schemes in the US, as farmer are concerned with 'practice profitability', rather than short-term subsidies (Norris and Batie 1987). They concluded that either subsidies were either not high enough or not offered on enough land.

Theoretically an optimal uniform input incentive equals the average of the expected marginal social costs created by the input use at each site, and adjustments to account for the average marginal impacts of *input substitution* on expected social costs and

[22] (Nowak 1987) identifies 15 constraints to adopting optimal technology.

[23] WQIP schemes offered incentives to undertake the following conservation measures: waste management systems, conservation cover, conservation tillage, split fertiliser applications, legume crediting, manure crediting filter strips, strip cropping, nutrient management, critical area planting, record keeping, pasture and hay land management and planned grazing systems among others (Higgins 1995).

profit levels (Shortle et al. 1998). These adjustments are required to counter substitution distortions and undesirable changes in the input mix (Eiswerth 1991). Thus the regulator would have to consider the likely management response to input/technology incentives, and pre-empt any detrimental substitutions by offering economic incentives or other measures to counter any detrimental characteristics of alternatives (Eiswerth 1991).

Although producers require no extra information or training, the regulator must have information about production, cost and runoff functions. More accurate information will result in more efficient corrective instruments. Although first-best instruments require perfect site-specific information[24], practical considerations (transaction costs, cost of gathering information, political and legal feasibility) would probably result in second-best solutions which only target a few important input and technology choices.

3.7.2.2 Cross-Compliance Mechanism

As was demonstrated in chapter 2, agriculture in the developed world receives immense financial support through direct subsidies and prices supports. Instead of offering farmer incentives and further subsidies to implement alternative agricultural practices (input and technology use), the regulator can withhold these subsidies unless farmers comply with polluting reducing measures. Thus cross-compliance ties the receipt of benefits from unrelated subsidy *packages* to some level of *environmental measures* e.g. implementation of specified technology or management practices.

The effectiveness of cross-compliance measures is limited because; 1) producers will only agree if the expected package benefits exceed the cost of implementing the environmental measures; 2) environmental benefit will only accrue to the extent to which those receiving program benefits (farmers) are contributing to water quality problems; and 3) Generally the benefits of compliance measures vary with economic

[24] Some management practices, such as the rate of nitrogen application, are difficult to measure without intensive, obtrusive and costly monitoring.

conditions (budgetary constraints and reduction in price supports) including fluctuations in world prices.

Whether cross-compliance is cost-effective depends on policy design, i.e. the extent to which compliance is based on performance. Generally cost-effective pollution abatement does not occur as programme incentives are unlikely to be distributed in a way which reflects contributions to water damages.

3.8 Marketable Pollution Permits

The converse of taxation is a system of quantity rationing through marketable pollution permits (MPP)[25], which specify a predetermined total level of emissions in a region. Rather than having some central authority establish a tax net, monitor performance, collect bills & create an adversarial relationship with polluters, MPPs rely on market mechanism and interaction between polluters. In legally selling the right to pollute, scarcity creates demand and supply forces that provide the incentive to trade. Since most pollution problems stem from externalities or ill-defined property rights, MPP allocate these rights and make them tradable.

Commentators (Hahn and Noll 1990) have established the following prerequisites for efficient MPP functioning: that the number of permits be fixed to enable permit price stabilising; least possible regulatory interference; that transaction costs be minimised; permits be storable to keep their worth in times of thin trading; penalties for violating permits be greater than permit price, therefore coercing the producers into market compliance; that producers be allowed to benefit from any profits from market transactions; and permits should only be expropriated in extreme circumstances to ensure reliability.

In an ideal world the regulator would presumably set permit quantity where marginal benefits and costs of pollution control cross, thereby ensuring the socially optimal reductions. However more realistically total permits are set with regard to an arbitrary target reduction in pollution (or standard). Under uncertainty however,

[25] Introduced by Crocker 1966 & Dales 1968.

depending on the relative slopes of curves, where the marginal benefit curve is steep (slight variation in emissions can potentially result in great damage), the permit performs better than taxes, but in the other extreme where the MB curve is flat, an emission tax is preferable. There are numerous discussions on regulatory possibilities under slope uncertainty and the best strategy (Baumol and Oates 1988; Hanley et al. 1997).

Permits are issued either freely, being allotted depending upon current emissions (*grandfathering*[26]) or are auctioned[27]. The initial allocation does not change the outcome as much as it alters the distribution of wealth. With auctioning, firms with high MAC would be willing to pay more than firms with low costs, and once markets are established high MACs to be buyers and low MACs to be sellers[28]. In essence each firm will compare its MAC schedule with the current market price and if price falls it will purchase more; so in effect the MAC curve for a firm is its demand curve for permits. In equilibrium, the market settles where each firm purchases permits at the level where it's MAC equals permit price or the cost-minimising solution. The efficiency of MPPs was first established by Montgomery in 1972. However Tietenberg's proof is somewhat more lucid (Montgomery 1972; Tietenberg 1994). There is no doubt that permit schemes offer superior incentives to innovate, in comparison to technological standards, simply because lowering abatement costs enables firms to spend less on permits (Downing and White 1986; Milliman and Prince 1989).

Suppose a is the ambient nitrate levels in the controlled catchment, given by:

$$a = \alpha + \sum_{i=1}^{n} (r_i - \theta_i) \qquad \text{(EQ-3.14)}$$

[26] If grandfathered there will be no resource transfer from the polluters *en bloc* , whereas this will be the case with auctions. Another strategy involves an allotment followed by an auction.
[27] Montgomery 1972, proved formally that the least-cost outcome is independent of the initial allocation of permits, however this depends on the circumstances i.e. transaction & information costs.
[28] Assuming of course, that initial allocation is not optimal i.e. the least-cost one.

where α are emissions from other natural sources, r_i are uncontrolled emissions from $i = 1 \ldots n$ polluting firms, θ_i are emission abatement by firm i, which costs the firm C_i in control costs:

$$C_i = C_i(\theta_i)$$

where, $C_i(\theta_i)$ is a continuous, twice differentiable function, with $C' > 0$ and $C'' > 0$. The regulatory standard is set at a_o. The agency's problem is to

$$\underset{\theta}{Min} \sum_{i=1}^{n} C_i(\theta_i) \tag{EQ-3.15}$$

subject to:

$$\alpha + \sum_{i=1}^{n} (r_i - \theta_i) \leq a_o \;,\; \theta_i \geq 0 \tag{EQ-3.16}$$

The Lagrangian is

$$L = \sum_{i=1}^{n} C_i(\theta_i) + \lambda \left(a_o - \alpha - \sum_{i=1}^{n} (r_i - \theta_i) \right) = 0 \tag{EQ-3.17}$$

Differentiating with respect to θ_i yields the following Kuhn- Tucker conditions for an optimum $\frac{\partial C_i(\theta_i)}{\partial \theta_i} - \lambda \geq 0$; $\theta_i \left(\frac{\partial C_i(\theta_i)}{\partial \theta_i} - \lambda \right) = 0$; $\alpha + \sum_{i=1}^{n} (r_i - \theta_i) \leq a_o$;

$$\lambda \left(\alpha + \sum_{i=1}^{n} (r_i - \theta_i) - a_o \right) = 0 ; \theta_i \geq 0; \quad \lambda \geq 0 \tag{EQ-3.18}$$

The above shows that λ is that shadow price of the pollution constraint. All firms' marginal abatement costs, given as $\frac{\partial C_i(\theta_i)}{\partial \theta_i}$, must equal λ. For a permit market to achieve this outcome, the regulator should issue a permit supply of $E^* = \sum_{i=1}^{n} (r_i - \theta_i)$, the permitted level of emissions. Under this emission permit system (EPS), dischargers will trade at a 1:1 rate. Suppose each firm is allotted an initial allocation of e_i^o where $\sum_{i=1}^{n} e_i^o = E^*$ and the price of p is initially arbitrarily set. Now each firm's problem is to

$$\underset{\theta_i}{Min}\, C_i(\theta_i) + p\left(r_i - \theta_i - e_i^o\right) \tag{EQ-3.19}$$

the solution being:

$$\frac{\partial C_i(\theta_i)}{\partial \theta_i} - p \geq 0\, ; \theta_i\left(\frac{\partial C_i(\theta_i)}{\partial \theta_i} - p\right) = 0\, ; \theta_i \geq 0 \tag{EQ-3.20}$$

In comparing these with the Kuhn-Tucker conditions, it is obvious that the least cost solution will be if price p is equal to λ, which will be the case in a competitive market.

3.8.1 MPPs and Non-point Pollution

Given NPS emissions as such cannot be directly measured[29] some proxy for their use i.e. estimated emissions from inputs & production process, must be controlled. A MPP system in nitrate inputs could be established. An interesting study modelled the use of MPPs for nitrate pollution the River Tyne (Northern England) catchment, and determined reductions in farm profits from two policies: a MPP system of estimated emissions (depending on land types and cropping patterns etc.) and MPP in nitrate inputs (Moxey and White 1994). As expected, since it is difficult to establish a simple relationship between nitrate use and ambient concentration because of varying transfer coefficients, *profits under a transferable pollution permit based on inputs were less*. But the administrative costs[30] associated with a market of estimated emissions would be greater if for each farm estimate emission were predicted. Alternatively, the amount of permits necessary to authorise a particular land use on some land type could be determined from estimated emissions. Once established the regulator would not have to repeat this every time trade took place (Pan & Hodge 1994). Although this is less efficient than an economic instrument, costs are reduced

[29] Nonpoint loadings are inherently stochastic because of their dependence on weather-related factors such as rainfall. (Shortle & Dunn 1986). Additionally their measurement is further complicated as emissions may take a long time to accumulate and reach surface waters.

[30] Including the cost of litigation, and disputes concerning emission estimates (English 1993)

as only monitoring land use and permit holding is required, and transaction costs would be lower.

3.8.2 MPP Permit Types

Ambient permits (APS) refer, not to source emissions, but to their effects at receptor points, thus effectively creating separate markets at each receptor point[31]. In theory a source whose emissions are 'more damaging to a particular receptor will have to purchase commensurably more emission entitlements from another source whose discharges contribute less per unit to pollutant concentrations at the receptor point.'(Baumol and Oates 1988). So each polluting firm must assemble a *portfolio of permits* from each affected receptor site, implying possibly high transaction costs.

Complications arise when fluctuations in environmental factors or changing spatial patterns of emissions generate 'hot spots', which do not coincide with designated hot spots. The nature and occurrence of hot spots depends on the type of water pollutant and how uniformly it mixes. To cover hot-spots, an APS would require a fine mesh of receptor points. However this is a costly solution which would further increases transaction costs. Potentially high transaction cost from trading in separate receptor markets and few traders might result in imperfect competition. Besides this, total emissions may rise, if one polluter sells permits allowing a fixed impact at a receptor point. Now the purchasing polluter may be located in such a way that he may actually increase emissions and yet the impact on the receptor would remain the same as before the trade. The possibility that APS can actually augment the increase in long-range transport of pollutants has been brought to attention (Atkinson and Tietenberg 1987a).

A simplified alternative is to divide the region into zones, and allow a one-for-one emission entitlement trading (EPS). Each source would be assigned to a zone and since polluters with somewhat varying dispersion coefficients are aggregated into the

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[31] Receptor points need not be fixed, and can be easily redefined to coincide with hot spots; nor do they have to be the points where actual pollution is monitored, as receptor are only reference points, and so regulators might determine receptor concentration based on some dispersion model etc.

same zone achieving a least cost solution is difficult. This is because one-for one trade ignores differences in the concentrations contributed by their respective emissions, i.e. '*the price of emissions to each polluter will not correspond accurately to the shadow price of the binding pollution constraint*' (Baumol and Oates 1988). The validity of this objection depends on whether the dispersion characteristics of emission within a zone vary.

In addition EPS requires the regulating authority to determine how many permits be assigned to each zone. The prerequisite for doing so involves a water model, source specific abatement costs and an emissions inventory so that the cost minimisation problem can be solved. The inability to do so properly, for whatever reason, would mean subsequent readjustment, or entering a market to purchase or confiscate permits. All of which are destabilising factors for polluters. To conclude, regulators face the persistent problem of periodic adjustments given new entrants, economics swings etc.

An ingenious compromise, termed the *pollution offset*, allows the *transfers of permits subject to the restriction that the transfer does not result in a violation of the environmental quality standard at any receptor point* (Krupnick et al. 1983). The rate at which emissions from one source can substitute for emissions from the other, whilst ensuring no change in pollutant concentration at the receptor is the *ratio of the sources' transfer coefficients*. Of course, ruling out one-for-one trading, if X discharges twice as much pollutant at a receptor than Y, then for X to double emissions effecting the receptor he must purchase two of Y's permits. Thus like an APS, mutually beneficial trades result in the least-cost solution[32], independent of the initial permit allocation. The authorities need not know the abatement cost functions and nor do they need to solve the cost-minimisation problem to determine initial allocation of permits (any allocation will suffice). However they obviously need to know the dispersion characteristics of emissions, in order to be able to declare official "exchange rates". Another benefit stems from lower transaction costs, as

[32] Baumol & Oates demonstrate the coincidence of the trading equilibrium with the least cost solution with the aid of simple diagram.

firms need not trade in a multitude of separate markets for each receptor site, rather they purchase directly from other sources.

Pollution offsets are further divided into the *non-degradation offset* which further restricts by disallowing all trade that would increase total emissions (Atkinson and Tietenberg 1982); or the *modified offset* which allows trades so long as neither the pre-trade quality level nor the target level, which ever is stricter is not violated (McGartland and Gates 1985). In latter work Atkinson and Tietenberg, conclude that there is no all-pervasive conclusion on the relative-effectiveness of either variants, however, they discredit the simple offset system (Atkinson and Tietenberg 1987a).

3.8.3 MPP and Market Structure

A potential problem may arise if there are only a few polluters (farmers) in a catchment, or if one is large enough to dictate permit price through its behaviour. This market imperfection will distort the ability of MPPs to be the least cost solution. By exerting monopoly power[33] a firm/farm can restrict the number of permits in the market and force up the price. Hence when it does sell, the cost of freeing up permits for sale is not given by its marginal abatement schedule. A formal analysis of firms and market behaviour under imperfect competition can be found in the literature (Hahn 1984; Misiolek and Elder 1989).

At the other extreme one firm/farm with monopsonistic power can reduce the price it pays for purchase, by buying fewer permits. This of course depends on market regulations and firm's MAC schedule. Thus, in buying too few, compared to the competitive outcome this firm will spend too much on abatement[34]. It is possible that un-competitive behaviour may be motivated by an attempt to increase permit prices for competitors, actual or potential (Misiolek and Elder 1989). Exclusionary behaviour may motive firms to distort permit price in order to prevent new entrants. Obviously such leverage is exerted by large firms with low MACs. In the case of a

[33] The extent of which depends on the price elasticity of demand (determines the permit price rise as more become available) and the slope of the MAC schedule (determines price of freeing up permits for sale).

monopsonist exclusionary behaviour will counter act cost-minimising behaviour and the distortion will not be severe, because a monopsonist is likely to exert influence by buying too few permits. However where a polluter exerts monopolistic influence i.e. by not selling permits, exclusionary behaviour will worsen the distortion.

3.8.4 MPP and Multiple Pollutants

In reality there are normally several agricultural pollutants (nitrogen, phosphorus, sediments) in waterways which jointly contribute to general habitat deterioration or fish mortality. If pollutants interact linearly, then there will be no significant alteration to theory. If two pollutants in acting together exert a greater influence than a linear sum of the two, synergism is said to hold[35]. Under a MPP system with linear damage functions from multiple pollutants, trading across firms would still result in the least cost outcome provided *permits were traded off at a rate equal to their relative contributions to the environmental quality indicator* (Hahn 1989a).

It has been also proven that where the damage function is non-linear and synergistic, permit trading alone does not represent the least cost outcome (Zylicz 1993). In fact a hybrid system of taxes and permits achieves the least cost solution only if the damage function is quadratic. When this is the case taxes will have to be firm specific and depend on emissions from that firm and their interaction with other pollutants. To the regulatory authority this is clearly a cumbersome and complicated procedure. Unfortunately such solutions do not apply when there are other non-linear forms.

3.8.5 MPP Problems

It is tempting to conclude that more use should be made of permits in dealing with agricultural water pollution. However in practice this has not been the case (Hanley et al. 1990). Where water bodies are concerned complex trading rules regarding non-uniformly mixed pollutants, non-linear synergistic interactions between pollutants, stochastic influences, high transaction costs, too few participating parties (lack of

[35] One such work studied the effect of a point source of cyanide on fish in the presence of other heavy metals (Beavis and Walker 1979).

market structure) and public perception (Hanley et al. 1990) can contribute to practical difficulties[36].

Permits have been simulated assuming multilateral, simultaneous, and perfect information (Krupnick 1986). Unfortunately in practice most trades[37] are bilateral, sequential, and often occur in the absence of information on the minimum compensation demanded (supply) or the maximum willingness to pay (demand price). The ramification of these restrictions (Atkinson and Tietenberg 1987b); and the implication of sequential trading (Klaasen and Frosund 1993) has been discussed in the literature. The following possible explanations for low levels of trading have been given (Munro 1995): management prefers abatement to permits, despite the associated profit loss; the belief that permit allocation in the next round depends on current holding, hence reluctance to sell; high transaction costs; and firms are unwilling to sell permits to their rivals.

Transaction costs arising from search & information; bargaining & decision; and monitoring/enforcement (usually borne by the regulating agency) can in some cases reduce trading and increase abatement costs. There is considerable empirical evidence on transaction costs in the literature (Atkinson and Tietenberg 1987b; Hahn and Hester 1989).

In considering the transaction costs from brokerage services, insurance fees etc. Stavins concludes that *'it is possible that in some circumstances the total cost of compliance (including transaction costs) of a tradable permit system could exceed (depending on initial allocation of permits) the costs of a uniform performance standard'*, (Stavins 1995).

Another implication of his analysis is that initial allocation of permits affects the final equilibrium when marginal transaction costs are non-constant. This implies that

[36] Other issues include whether trade can be limited to polluters only or may environmental groups purchase and retire permits as a means of reducing pollutants.

[37] There are numerous types of transactions ranging from temporary rentals and long-term leases to permanent transfers.

when transaction costs are present and prohibitive, operating an efficient market means distributing permits initially so that the sum of control costs and transaction costs are minimised. He attributes the poor popularity of ambient or concentration permit trading, exposure trading (Roumasset and Smith 1990) and risk trading to higher transaction costs; and advocates that government should take on brokerage responsibilities.

3.8.6 MPP Point-Nonpoint Trading

An interesting concept is the proposal allowing trade between point and non-point sources. Point-nonpoint trading would allow point sources to sponsor nonpoint source controls, rather than have them install further controls of their own. If nonpoint source loadings are significant and the marginal costs of their control are lower than for additional point source controls, ambient water quality could be achieved at significantly lower costs. Nor does it lessen incentives to innovate and invest in control technologies, rather the opposite.

In acknowledging that agriculture is the single largest contributor of nonpoint source pollution research in the US investigated the feasibility & potential for point/nonpoint trading in coastal watershed (Crutchfield et al. 1994). The research concluded that although sound in theory, successful implementation of market-based approaches depends in part on whether practical circumstances support the functioning of such a market.

Point/nonpoint trading is a 'bubble' concept applied to watershed management. A 'bubble' sums loadings for all area sources and allows adjustment of the level of control applied to each source so long as total loadings do not exceed a target level. In America, where there has been some limited application, PS-NPS trading has allowed granting publicly owned treatment plants and industrial point sources the option of bringing agricultural and urban NPSs under control rather than requiring more advanced treatment technologies at point sources. Normally the permit base is point source polluters trading emission allowances for allowances based on a) expected runoff by nonpoint or b) input use by nonpoint sources.

It is probable that there is a limit to such trading (some optimal trading ratio[38], subject to ecological constraints) for not all agricultural run-off and industrial pollutants discharged are similar in effect. A model to determine the optimal trading ratio found it was dependent on the relative costs of enforcing point versus nonpoint reductions and on the uncertainty associated with nonpoint loadings (Malik et al. 1993). This uncertainty does not imply a lower bound for the optimal trading ratio. The effect of uncertainty depends on source of uncertainty and on the curvature of the damage function. The effect of stochastic nonpoint loadings can be opposite to that of imperfect information about the effectiveness of nonpoint source controls. In both cases, the direction of the effect depends on whether the damage function is concave or convex in nonpoint loadings. Thus uncertainty does not imply that the trading ratio will be greater than one. But for all this to be calculated the regulator needs to know the damage function, and abatement costs to determine expected marginal enforcement costs. It must be noted that these results were for two sources only, one can imagine the complexity of allowing multiple, heterogeneous point and nonpoint sources with growth and stock effects of pollutants.

In fact there are numerous difficulties with NPS controls which make PS payments to induce farmers to adopt conservative practices insufficient to ensure compliance or that targets are met. Firstly loadings cannot be monitored or measured directly at source (some input proxy[39] is employed), since nutrients, sediments and other chemicals enter water streams over a dispersed area. Secondly, there are imperfect relationships between loadings and farm-level decisions dealing with input use, land management, and their fate- transport mechanisms. A third complicating factor is the random nature of episodic events i.e. wind, rainfall, and temperature. So regulating officials find it difficult to determine whether failure to meet water quality goals is

[38] Trading ratio specifies the rate at which nonpoint source abatement can be substituted for point source abatement. Nonpoint allowances are generally not 1:1, due to the random nature of nonpoint loadings and the heterogeneous nature of NPS contributions to pollution (Ribaudo et al. 1999). A uniform trading ratio equal to the price of an point-source emission allowance relative to the price of an expected runoff allowance defines the number of point emissions allowances that must be traded for one unit of expected runoff.

[39] Proxies include quasi-fixed inputs such as fertilizer, pesticides, irrigation water, management practices, and conservation investments etc.

due to failure of individual sources to correctly implement control strategies or due to undesirable states of nature. To make matters worse, enforcement cannot be based on attainment of ambient water quality targets, because it often takes several years of monitoring to detect and attribute water quality changes to NPs controls; and monitoring in short term has to focus on land use practices.

Because of the aforementioned reasons PS would generally sponsor a larger reduction via the NPs than the reduction they aim to avoid. Another consideration is the number of participants, as too few participants would lead to inefficiencies. Again, as under any kind of permit trading, single or a few large PS acting together would could behave as a monopsonist and purchase less NPS pollutant reduction than is socially optimal[40]. Actually the reverse is troublesome as well, as too many participants would raise the co-ordinating and bargaining costs. Where transaction costs are significant, the potential benefits are limited. Similarly significant NPS loading must exist because the objective of PS is to avoid abatement technology upgrades, which are often discrete or 'lumpy' because of capital requirements.

When trading point-source emissions with nonpoint variable production inputs, the efficient trading ratio is defined as the marginal rate of substitution of emissions for input use such that expected damages and pre-permit profits are held constant (Shortle and Abler 1997). Ideally with n farm sites and m inputs influencing diffuse, n x m markets (trading ratios) are required to achieve efficiency. However given transaction costs of monitoring, administrating markets and enforcing permit compliance second-best allocation could be obtained by allowing trading at uniform rates and by limiting the number of tradable inputs[41]. Price uniformity will however reduce the cost-effectiveness of pollution control because it eliminates potential gains from different treatment of polluters according to their relative ambient impact.

Trading ratios in excess of one implies high nonpoint control cost relative to point source control and thus a marginal preference for point source reductions. The

[40] Here regulatory officials would have to intervene by adjusting the substitution rate, or trading ratio.
[41] Which inputs are traded would depend upon: the nature of any resulting substitution effects, monitoring/enforcement costs, correlation with environmental quality (Ribaudo et al. 1999).

reverse is true of ratios less than one. The magnitude of an optimal trading ratio cannot be determined a priori. It is a site-specific empirical issue involving a) relative marginal impact of point and nonpoint sources, b) heterogeneity of sources, c) degree of environmental impacts, d) transaction costs of each type of emission control, and most importantly, f) correlation between environmental and cost relationships (Shortle 1987; Malik et al. 1993). The trading ratio will be negative for inputs which reduce emissions, and the regulator might define minimum required input use, as a firm might tend to under use such inputs (Shortle and Horan 2001).

There are three major differences between E-I (point emissions for NPS inputs) and E-EL (point emissions for NPS loadings) trading, 1) E-EL trading conveys more site-specific information to producers about the environmental impacts of their choices, 2) E-I trading allows for differential targeting of inputs (Shortle and Horan 2001), 3) theoretically, transaction costs aside, E-I systems are more efficient because they are better at managing the natural variability of NPS loads (Shortle and Abler 1997). It is noteworthy to remember input substitution i.e. the regulator must be aware that although permit trading may require an increase in pollution-reducing inputs it may also result in increased demand for pollution-increasing inputs and further environmental damage (Shortle and Horan 2001).

Pollution trading has gained popularity in the US as a cost-effective approach than the traditional 'command and control' approach. Point/Nonpoint trading programmes have been set up in several US water bodies notably Dillion and Cherry Creek Reservoirs in Colorado, and Tar-Pamlico basin in Carolina with varying success (Hogg and Holloway 1991; Malik et al. 1994). Most of them permit point-source polluters purchase emission allowances from NPS farmers through the installation of best-management practices (BMP's) and the development of nutrient management plans. No markets currently exist for trading allowances based on nonpoint inputs.

3.9 Other NPS Issues

The following section will briefly outline the other important diffuse pollution issues such as mixed instruments, non-uniformly mixed pollutants, dynamic NPS pollution, stock and multiple pollutants and product taxation. It must be remembered that *this section refers to NPS pollutants in general not specifically diffuse nitrate pollution*; the chemical and physical properties of NPS pollutants vary considerably.

3.9.1 Mixed Instruments

Economic instruments applied to a single compliance base are efficient in a first-best world under highly restrictive and unrealistic conditions. Numerous authors have considered the use of mixed instrument incentives to address the informational challenges in NPS control (Braden and Segerson 1993).

One proposal involved mixing input taxes with liability rules in a bid to overcome the real world implementation problems of each if implemented separately (Braden and Segerson 1993). However the authors warn that *a prior* it is not possible to predict whether the combination of two instruments is more efficient than if either was implemented on its own. In an investigation of a combined product and emission tax it was concluded that the policy mix is optimal when a) fixed costs of emission monitoring were low, and b) the presence of initially low marginal costs which increase with the monitoring effort.

Similarly it has also been demonstrated that a combination of input and ambient taxes can give a first-best solution even when polluters are risk averse, something which cannot be achieved with ambient taxes alone (Horan et al. 1998). Xepapadeas investigated an ingenious method to circumvent high emission monitoring costs by combining emission and ambient taxes to a policy which would lead polluters to reveal all or part of their emissions (Xepapadeas 1995).

The result is similar with that of earlier work which investigated the complimentary interaction between *ex post* negligence liability and *ex ante* regulation i.e. pigouvian taxation (Kolstad et al. 1990). The research demonstrated that given uncertainty a policy mix of the two is likely to be more efficient than the exclusive use of one. This efficiency gain is especially so if the injured party's marginal cost of precautionary measures is significant at the social optimum or if there is considerable uncertainty in setting the legal safety standard. If this is the case then an efficient outcome can be expected provided a pigouvian tax is levied alongside a less precautionary safety standard which is below what would be used if the liability rule was used alone. Later research considered and confirmed efficiency gains in employing the same two tax instruments (ambient and pigouvian) jointly when co-operation between producers is likely (Millock and Salanie 1997).

3.9.2 Non-Uniformly Mixed Pollutants

Obviously the assumption that NPS emissions from all sources are uniformly mixed together i.e. emissions from one source have the same marginal impact on ambient quality as those from other sources, is not realistic. The fact that some pollutants have short natural half lives[42] is another factor determining impact. Factors such as seasonal flow fluctuations, rainfall, temperature, etc. all contribute to this difference. So the same quantity and concentration of pollutants discharged closer to a monitoring point will have a significantly greater impact or *technical coefficient*[43] than one discharged further upstream. Although a single tax would achieve the desired reduction, it would not be at a minimum cost (Tietenberg 1994). Thus for mixed pollutants tax rates should vary depending on their marginal impacts on ambient water quality i.e. transfer coefficients.

One could either calculate taxes on the basis of transfer coefficients for the most polluted monitoring point or have separate tax rates for each monitoring point, which are then adjusted for each firm according to its transfer coefficient at the receptor site (Tietenberg 1973). To illustrate, suppose a source B whose transfer coefficient is twice that of another A (be it for spatial location differences or whatever); the effluent tax paid by B must be twice as that paid by A to ensure that the marginal abatement cost of B would be twice that of A's. But, note, the damage reduction *per dollar spent in reducing emissions* would be equalised across the two sources[44].

[42] I.e. they undergo natural degradation and re-aeration, depending on the river's assimilative capacity and the distance between discharge and monitoring point etc.,

[43] For a detailed account of environmental modelling and the construction of a transfer coefficient matrix consult O' Neil et al. (O'Neil et al. 1983).

[44] Thus for every firm there is a unique shadow price or tax rate at each monitoring point.

This theoretical idea has no place and would be "administratively difficult at best and politically unfeasible at worst" (Tietenberg 1974);he goes on to outline a location-determined or *zonal* tax system where the administrator divides a territory into separate zones depending on relative impact i.e. grouping together sources whose emissions have similar effects on ambient quality levels. Here tax rates vary across zones but not within them. A similar zonal treatment with marketable pollution permits has been investigated (Moxey and White 1994). Details of this study along with other issues, such as site-specific information (targeted policies) are discussed in detail in the following chapter on empirical works.

3.9.3 Dynamic Non-point Pollution

So far the discussion has involved introducing a *static* charge to remedy the moral hazard problems associated with imperfect monitoring. However such schemes ignore the effects of 'a dynamic process of pollutant accumulation on individual behaviour when it is imposed, through some incentive scheme, as a restriction on the dischargers' profit-maximisation problem' (Xepapadeas 1991; Xepapadeas 1992b). This leads to sub-optimal inefficiencies in the long run, as water quality at each point in time depends on pollutant *stock.* Xepapadeas designed an inter-temporal incentive scheme to induce dischargers to comply with a socially desirable long run equilibrium. The charge depends on the pollutant's shadow cost, natural decay rate, the discount rate & the parameters associated with the model's information structures. Obviously when firms follow feedback strategies i.e., when they condition emissions on the currently observed pollutant levels, the required charged is higher than when open-loop strategies are followed.

3.9.4 Stock & Multiple pollutants

Often ground water is affected by *stock* pollutants, i.e. those which accumulate through time with continued emissions and which might undergo a natural rate of decay. Initial economics work on stock pollution was pioneered as early as 1972 (Plourde 1972). A later study modelled the build up of aldicarb in Long Island (US) where it had been banned for twelve years (Conrad and Oslon 1992). They used an equation to construct a possible *time path for stock pollutants*, something which must be done to levy an efficient tax; they estimated the social value of damage and concluded the build up was a result of setting the application of aldicarb limit too high.

Most economic research only considers single pollutants, and not their combined effects. In practice, unfortunately, undesirable environmental effects are normally brought about by the joint presence of interacting toxins. A study of economic incentives to control the accumulation of green house gases involved setting tax rates for each contributing pollutant and assumed a finite assimilative capacity in each time period (Michaelis 1992). The research concluded that relative tax rates between pollutants depend on their relative damage and dispersion co-efficients, and that for efficiency taxes must grow at the discount rate, adjusted for the decay rate of pollutants. In accordance, tax rates should rise over time as scarce overall assimilative capacity diminishes.

3.9.5 Product Tax

A product tax lowers the price for polluting activities, and should in theory reduce the profit maximising fertiliser application. The first order conditions for profit maximisation show that the marginal product equal the ratio of input to output prices, thus halving the product price should have the same effect in theory as doubling fertiliser price, provided the cultivated area and product mix do not change. However in reality crops differ with respect to product prices and nitrogen requirements (Sumelius 1994a).

In comparison with taxes and subsidies, product taxes do not discriminate between polluting and non polluting industries. Additionally it must be remembered that most economic incentives normally encourage investment and research in less polluting technologies, this is not the case with product taxation. Product taxation would also be fraught with political problems.

The following section on property rights and the polluter pays principle will form a basis for discussing subsidies in the next section.

3.10 Property Rights and the Polluter Pays Principle

The choice of instrument in practice is determined by factors other than efficiency, such as political feasibility and ethical considerations. A subsidy (i.e. opposite of taxation whereby polluters are paid to reduce emissions) to farmers to reduce pollution implicitly affirms their right to pollute and the view that the public must pay for non-contaminated water. The alternative view that polluters must 'pay' for their toxic discharges to the environment is supported by levies. The polluter pays principle (PPP) has been the regulatory policy shaping point source discharges (Ribaudo et al. 1999). Interestingly if society does own the environment then the extra burden on polluters under pigouvian taxation i.e. the difference between the total tax burden and the total external costs can be considered the *environmental rent*[45] (Spulber 1985; Spulber 1989).

Existing property rights of farmers in developed countries allows them to grow what they choose, within bounds, in any amount. This is the result of the prioritisation of food production over the environment, in earlier times when environmental degradation was not threatening (Bromley and Hodge 1990). Such entitlements are not cast-iron and subject to change as society's perceptions and priorities evolve (Segerson 1996).

The polluter pays principle[46] (PPP) was incorporated in EC law in 1987, although there are references to it in the first EC Action Programme on the Environment in 1973. The political and social considerations which have prevented the implementation of the PPP in agriculture are numerous (Baldock and Lowe 1996; Tobey and Smets 1996). The meaning of the word 'polluter' has also changed over time and it has come to mean someone who *directly or indirectly damages the*

[45] Economic rent is the return on a commodity in excess of the minimum required to bring forth its services, or the excess return to an input, i.e. the difference between payment for resource use and the lowest the owner would have been willing to accept.

[46] The polluter pays principle was established by the OECD, i.e. those who use society's environmental resources must compensate the owners (public) for any degradation. It was first widely discussed and brought to the public's attention at the United Nations Conference on Environment and Development held in Rio de Janeiro of Brazil in June 1992. The principle was endorsed by all the attending representatives of the countries and nations. However it has taken a long time for countries to implement it.

environment. Over the years there have been numerous disputes regarding the PPP's practical interpretation such as the relative role of regulation versus economic instruments in implementation; and whether polluters have to pay the full control costs and/or restoration measures. These have largely arisen due to the flexibility granted to member states in interpreting directives.

Recent consultations and the Commission's 2000 Environmental Liability White Paper (COM(2000)66) refers to the possibility introducing an EU liability regime for environmental damage. In fact the clearest signal yet has been the proposed sixth Environmental Action Programme (COM(2001)31) – *Environmental 2010: Our Future, Our Choice* – proposes the following commitment:

' *To promote the polluter pays principle, through the use of market based instruments, including the use of emissions trading, environmental taxes, charges and subsidies, to internalise the negative as well as the positive impacts on the environment*'. – (proposed Article 3(3)).

Such a proposal will integrate environmental considerations in 'sectoral decision making' (proposed article 6) especially both future agricultural and water policies.

3.11 Subsidies

In a departure from the stick or rod tax approach, subsidies reward firms for better environmental practices i.e. decrease the cost of pollution mitigation. The choice of taxation over subsidies (under the PPP) is not just a *normative decision*, as subsidies are fraught with economic distortions. Although in violation of the polluter pays principle, in the short run subsidies and taxes are equal. Essentially under a subsidy producers select output to maximise net profits where the marginal benefit i.e. price, equals the marginal private cost and the marginal opportunity cost of lost subsidy. Every unit of output results in a lost unit of the subsidy[47]. Undoubtedly there is an incentive to reduce an individual's output to the socially optimum.

However, subsidy payments are made relative to a specified benchmark, e.g. a subsidy on fertiliser use might be based on a reduction in fertiliser application from a specific level. The specification of such benchmarks may create perverse incentives, as establishing site-specific or catchment level benchmarks at current discharge levels would penalise farms which have already committed themselves to pollution abatement (Baumol and Oates 1988).

Nonetheless subsidies encourage entry by new firms and raise overall output. Subsidies inevitably sustain firms that are inefficient and unprofitable, those that would have exited in the absence of financial support (Lewis 1994). In essence this is an information rent as the regulatory authority cannot differentiate between high and low profitable industries. If the correct number of firms is desired then in the long run firms should pay not only the marginal damage but also the total cost arising from waste emissions (Spulber 1985). However it is possible to conceive of subsidies as not affecting entry/exit decisions (Segerson 1990), e.g. subsidies paid to particular land parcels, such as filter strips, may be capitalised into land values in the long run. Remember, excessive entries are not possible because the area of land next to waterways is fixed.

Another issue with subsidies is their use with abating inputs, such as the use of leakage reducing crops. As an abating input subsidy creates an incentive to use more of it, normal first order conditions do not apply. It has been shown that when an abating input enters the production function, optimality cannot be derived by imposing negative or positive taxes on that input (Weinberg 1991). Some have attempted to justify subsidies by arguing they mitigate the economic impact of environmental regulation by helping firms meet compliance costs. Given the current production subsidies and price supports to farmers in the UK subsidising agricultural emissions would merely add to the tax payer's burden.

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[47] For a formal proof consult Hanley et al, 1997.

3.12 Non-Economic Approaches

The term 'non-economic' approach is used to group both farmer *education* and regulatory *standards* i.e. approaches which require or mandate that producers behave in a specific manner. Neither education nor regulatory standards directly use the price incentive to induce producers to account for externalities.

3.12.1 Education

Education refers to providing farmers knowledge about how their production decisions affect the environment (pollution generation relationships) and about new less polluting technologies (which use polluting inputs efficiently) and management (Bosh et al. 1995).

Education is often considered a 'win-win' solution, as it encourages producers to adopt practices (nutrient, tillage, irrigation water management (Ervin 1995)) which not only increase production but also water quality. It is popular because:

a) It is less costly to implement than a cost-sharing program (Ribaudo et al. 1999).
b) The infra-structure for implementing an information dissemination program is normally present in most farming communities.
c) It is a benign form of intervention, i.e. not mandatory and relies on farmer's good-will.
d) Generally it keeps with the public's predominantly environmental friendly idea of farming.

If education is to be a cost-effective NPS pollution control measure then utility maximisation requires that alternative practices be more profitable than conventional practices, or that producers value cleaner water enough to potentially lower profits. Additionally, altruism or stewardship motives will only result in change if producers believe there is an environmental problem and that their actions contribute to it. However, surveys suggest that most farmer do not admit this (Lichtenberg and Lessley 1992).

Evidence suggests that in the adoption of new practices through education producers are mainly concerned with net returns and not altruistic concerns for the local environment (Abler and Shortle 1991). Essentially by providing education the regulator attempts to bring about voluntary control measures. In a study of voluntary nitrogen fertiliser extensification in Finland the authors concluded that the majority of farmers were unable to save in production costs by reducing fertiliser expenditure as marginal returns are significantly higher than marginal costs (Sumelius 1994b). Generally economists are sceptical of using moral suasion alone to control environmental externalities.

Given the reduction of price supports under the CAP reforms and trade liberalisation, it is unlikely that farmers will adopt costly or risky pollution control measures for altruistic reasons alone.

3.12.2 Regulatory Standards

Legal mandates requiring farmers to behave in a specified manner i.e. regarding input use or a particular technology are referred to command and control measures (CAC) or regulatory standards. These can be applied to either producer actions (design) or the outcome of their actions (performance)[48].

3.12.2.1 Performance Standards

CAC performance bases need to be stated in terms of the moments of the performance base (mean, variance etc.) or the probability of attainment (e.g. ambient levels should not exceed the standard more than 90% of the time) because the performance bases (runoff, ambient concentrations or damages) cannot be controlled deterministically (Ribaudo et al. 1999). Performance based CAC measures have several disadvantages:

a) Given that the bases are stated as moments of the pollutants distribution or probability of attainment monitoring would have to occur over a period of time to determine the sample distribution of the base.

[48] The choice of CAC base for point sources is clearly emissions as they can be easily monitored (Baumol and Oates 1988).

b) The required monitoring timeframe may be significant, e.g. the movement of nitrates to groundwater can take numerous decades. Thus it may take years to before the regulator can verify compliance
c) The informational requirements are immense and unrealistic, i.e. polluter must understand how their input and technology choice and those of *others* determines runoff and contribution to ambient levels.
d) All polluters must have identical expectations about random processes.

The above limitations make performance based CAC measures unrealistic policy tools.

3.12.2.2 Design standards

Design standards can be formulated to place restrictions on expected runoff, inputs and technology. Here expected runoff is calculated by monitoring farmers input and technology decisions and then feeding them into a biophysical simulation model tailored to the catchment. This allows producers to use any private knowledge they might have about input and technology use, so long as it can be captured by the simulation model. Under such a system a cost-effective solution is attainable[49] and optimal entry/exit in achieved by setting the standard such that there are more economic benefits in retiring extra-marginal land. However the following caveats apply:

a) An expected runoff standard will only be effective if farmers comprehend how their production and pollution abatement decisions influence their emissions in the eyes of the regulator, i.e. they must have the same emission generation expectations as the regulator.
b) CAC incentives based on expected runoff are subject to efficient outcomes under restrictive conditions (Ribaudo et al. 1999) just like economic measures on the same base.
c) There may be legal problems with basing standards on the regulator agencies expectation of runoff and not actual runoff.

d) High transaction costs in reality (associated with monitoring and enforcement) may lead to the imposition of a uniform standard (ignoring site specific marginal impacts or land heterogeneity) resulting to a significant loss in efficiency.

3.12.2.3 Technology-based Effluent Standards

A technology-based effluent standard (TBES), or a design standard, is an effluent/emission standard set at the level of emissions that a source would produce if it were employing a particular type of abatement technology. Farms can control emissions by altering their choice of tillage, irrigation technology, management practices etc. Besides achieving a desired emission any combination of the previously mentioned will alter the cost functions of firms, indeed this may constitute heavy investment and hence affect the firm structure, profits, employment and competitive edge.

For each category of polluting source the regulating authority sets standards after considering costs and emissions from a representative farm employing the technology. The criterion in considering each technological option [50] is either the *best practicable technology*, or a more stringent *best available technology*. As is apparent the word 'practicable', is open to interpretation and the authority's discretion, i.e. prone to political or pressure group influence. This lack of transparency allows the agency to set standards, a time consuming and expensive procedure, depending on their interpretation of available, practicable & economic feasibility. Even if complex regulatory TBES could be easily set, they are efficiency considerations as most standards are set by reference to available technology and not to ambient water quality. Applying uniform standards irrespective of the spatial location of urban population density or sensitive ecosystem is not efficient as potential damage is variable. In other words the reduced emissions are not worth the cost, as when implemented, complying sources will not have the same marginal

[49] Proof of an efficient and cost-effective solution of under command and control measures can be found in literature (Hanley et al. 1997; Ribaudo et al. 1999).
[50] Each technological option refers to a particular collection of technical management, input operation procedures etc.

abatement costs. Additionally TBESs effectively reduce incentives to innovate, as the emission standard is linked to a particular technology

Other difficulties remain as there is a difference between *initial* and *continued* compliance. The fact that a farm has installed certain technology or stated its commitment to follow certain procedures equipment does not necessarily mean that this management intiative will be efficiently implemented in the future. Depending on the size of operating costs there may exist incentives to renege. The important point is that neither design nor performance standards achieve cost efficiency when marginal abatement costs vary across firms, except by chance.

A technology and input based CAC measure is relatively simple to introduce, the regulator simply mandates the technology and input levels which yield the greatest level of expected net benefits for society. The regulator should set controls which account for input substitution[51] and prevent production on extra-marginal land. Again transaction cost considerations will result in a uniform standard with poorer allocative efficiency than site-specific standards. As with uniform taxation, uniform standards result in low abatement cost farms using more polluting generating inputs and less polluting abating inputs than is efficient. The opposite is true of firms with high pollution abatement costs. Thus the disadvantages of using CAC measures include:

a) Unlike under uniform taxation, marginal per acre profits are not equated across farms under uniform standards.
b) They are not self-adjusting under changing economic or environmental conditions. The regulator must alter the standard as required.
c) As they do not provide an incentive to use the socially optimum amount of inputs and technology, farmer decisions must be monitored at all times to ensure compliance. This can be extremely costly.

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[51] placing mandatory controls on easily observed inputs may result in substitution distortions and even result in more pollution generation (Eisworth 1991).

d) They leave producers with little freedom regarding their production and pollution control choices, thus limiting their ability to utilise their unique site-specific knowledge.

In general CAC control measures are inflexible as they limit the ability of farmers to use their knowledge and do not provide incentives to comply with the socially optimal outcomes.

3.13 Conclusion

This chapter discussed the characteristics of NPS nitrate pollution as an economic externality. It also stated the NPS pollution policy first-best efficiency and second-best cost-effective conditions to ensure optimal regulation. The relative performance of economic (performance and design based instruments) and non-economic approaches (education) to regulating NPS nitrate pollution and their subcategories have been discussed. The major characteristics of each regulatory policy was detailed and debated on the basis of its 1) relative efficiency, 2) informational requirements, 3) relative complexity, 4) potential administrative and enforcement costs (i.e. transaction costs), and their 5) flexibility relative to changes in the economic and environmental conditions.

Performance based measures are generally infeasible at present because of the difficulty of observing nonpoint-source emissions and the information requirements placed on producers. Overall it can be concluded that it is the characteristics of NPS pollution (i.e. heterogeneous, stochastic etc.) and the practical considerations of second-best policies (due to transaction costs and political feasibility etc.) which favour the adoption of multiple instrument regulation policy.

Given the numerous instruments available and the site specific nature of NPS nitrate pollution it is apparent that the choice of regulatory policy should only be decided after an empirical analysis of the catchment. The next chapter details empirical studies to regulate NPS pollution in the economic literature at both the farm and catchment level.

Appendix[52] 3.1

Cost-Effective Solution Based on Mean Ambient Standard

The regulator's problem can be written as:

$$\underset{x_{ij}, A_i, n}{Max}\ J = \sum_{i=1}^{n} \pi_i (x_i, A_i) \qquad \text{(EQ- 6)}$$

subject to:

$$E\{a\} \leq a_o \qquad \text{(EQ-7)}$$

where a_o is an exogenously specified ambient standard, and the Lagrangian is:

$$L = \sum_{i=1}^{n} \pi_i (x_i, A_i) + \lambda [a_o - E\{a\}] \qquad \text{(EQ-8)}$$

where λ is the Lagrangian multiplier. Assuming an interior solution the first-order conditions with respect to input use and number of sites are:

$$\frac{\partial L}{\partial x_{ij}} = \frac{\partial \pi_i}{\partial x_{ij}} - \lambda E\left\{ D'(a) \frac{\partial \alpha}{\partial r_i} \frac{\partial r_i}{\partial x_{ij}} \right\} = 0 \ \ \forall i, j \qquad \text{(EQ-9)}$$

$$\frac{\Delta L}{\Delta n} \approx \pi_n - \lambda E\{\Delta a\} \approx 0 \qquad \text{(EQ-10)}$$

where $\Delta a = a(r_1, ..., r_n, W) - a(r_1, ..., r_{n-1}, W)$. The shadow value λ is the value of the optimal tax/subsidy rate when producers and regulator have the same expectations regarding the nonpoint process. Finally the optimal technology vector, A^*, is determined by solving for an optimal allocation for each possible value of A and comparing aggregate profits. The optimal technology vector satisfies the condition:

[52] *Note: The notation in this appendix leads on from the first-best solution listed in the main text of chapter 3.*

$$L\left(A^*\right) - L\left(A'\right) \geq 0 \forall A' \qquad \text{(EQ-11)}$$

this implies that the following must hold:

$$\begin{aligned}\pi_i\left(x_i\left(A_i^*\right), A_i^*\right) - \pi_i\left(x_i\left(A_i'\right), A_i'\right) \geq \lambda E\left\{a\left(r_1^*, \ldots, r_1^*, \ldots, r_n^*, W\right)\right\} \\ - \lambda E\left\{a\left(r_1^*, \ldots, r_{i-1}^*, r_i', r_{i+1}^*, \ldots r_n^*, W\right)\right\}\end{aligned} \quad \forall i, \forall A_i' \qquad \text{(EQ-12)}$$

Appendix[53] 3.2

Cost-Effective Solution Based on Mean Runoff Standards

The Lagrangian corresponding to the maximisation of eq-1 subject to $E\{r_i\} \le r_{io} \forall i$ is:

$$L = \sum_{i-1}^{n} \pi_i(x_i, A_i) + \sum_{i-1}^{n} \lambda[r_{i0} - E\{r_i\}] \qquad \text{(EQ-13)}$$

where, r_{io} is an exogenously specified runoff standard and λ_i is the Lagrangian multiplier for the ith runoff constraint. Assuming an interior solution the first-order conditions with respect to input use and number of sites are:

$$\frac{\partial JL}{\partial x_{ij}} = \frac{\partial \pi_i}{\partial x_{ij}} - \lambda_i E\left\{ \frac{\partial r_i}{\partial x_{ij}} \right\} = 0 \ \forall i, j \qquad \text{(EQ -14)}$$

$$\frac{\Delta L}{\Delta n} \approx \pi_n - \lambda_n [r_{n0} - E\{r_n\}] \approx 0 \qquad \text{(EQ- 15)}$$

The shadow values λ_i equal the optimal tax/subsidy rates when farmers and regulators have the same expectation about the NPS process. EQ -14, is a condition which implies the same conditionality as EQ – 2, however marginal costs are expressed as a constraint as opposed to actual damages. Similarly, constraint EQ -15 reduces to a zero profit condition for the marginal site.

As with an ambient standard the following two conditions regarding the optimal technology vector, A^*, are determined by solving for an optimal allocation for each possible value of A and comparing aggregate profits.

[53] *Note: The notion in this appendix leads on from the first-best solution listed in the main text of chapter 3.*

$$L(A^{*}) - L(A^{'}) \geq 0 \forall A^{'} \qquad \text{(EQ- 16)}$$

$$\begin{aligned} &\pi_i\left(x_i\left(A_i^{*}\right), A_i^{*}\right) - \pi_i\left(x_i\left(A_i^{'}\right), A_i^{'}\right) \geq \\ &\lambda_i E\left\{r_i^{*}\right\} - \lambda i E\left\{r_i^{'}\right\} \forall i, \forall A_i^{'} \end{aligned} \qquad \text{(EQ- 17)}$$

Appendix[54] 3.3

Cost-Effective Solution based on Input Use

Input standards may be defined in terms of either site-specific input use or aggregate input use within a catchment. The former case is considered. Let z_i denote the $(m' - 1)$ vector of inputs for which there are standards, and let y_i be the $([m - m']x1)$ vector of inputs for which there are no regulator restrictions (i.e. standards). Thus the inputs on which standards apply are defined as:

$$z_{ij} \leq \bar{z}_{ij} \forall i, j \qquad \text{(EQ – 18)}$$

where $\bar{z}_{ij}$ is the standard limit on the *j*th input on the *i*th site. These standards may be either stated in absolute terms or as probabilistic constraints. The Lagrangian which corresponds to the maximisation of EQ – 6 subject to EQ – 18 will be:

$$L = \sum_{i-1}^{n} \pi_i(z_i, y_i, A_i) + \sum_{i-1}^{n} \sum_{j=1}^{m'} \lambda_{ij}[z_{i0} - z_i] \qquad \text{(EQ – 19)}$$

where λ_{ij} is the Lagrangian multiplier for the *j*th input constraint for the *i*th site. Assuming an interior solution, the first order solutions with respect to input use and number of sites will be:

$$\frac{\partial L}{\partial z_{ij}} = \frac{\partial \pi_i}{\partial z_{ij}} - \lambda_{ij} = 0 \; \forall i, j \qquad \text{(EQ – 20)}$$

[54] *Note: The notion in this appendix leads on from the first-best solution listed in the main text of chapter 3.*

$$\frac{\partial L}{\partial y_{ij}} = \frac{\partial \pi_i}{\partial y_{ij}} = 0 \ \forall i, j \qquad \text{(EQ – 21)}$$

$$\frac{\Delta L}{\Delta n} \approx \pi_n - \lambda_n \left[r_{n0} - E\{r_n\} \right] \approx 0 \qquad \text{(EQ – 22)}$$

the shadow values λ_{ij} represent the optimal incentive rates for input use which should result in social welfare maximisation. EQ -20, is a conditions which implies the same conditionality as EQ – 2, however marginal costs are expressed as a constraint as opposed to actual damages.

$$\pi_i\left(x_i\left(A_i^*\right), A_i^*\right) - \pi_i\left(x_i\left(A_i^{'}\right), A_i^{'}\right) \geq$$
$$\sum_{j=1}^{m} \left[\lambda_{ij}\left(A_i^*\right)x_{ij}\left(A_i^*\right) - \lambda_{ij}\left(A_i^{'}\right)x_{ij}\left(A_i^{'}\right)\right] \forall i, \forall A_i^{'}$$

Chapter 4
Empirical Studies of NPS Pollution Control

4.1 Introduction

This chapter reviews empirical studies of NPS pollution control. The previous chapter outlined various theoretical economic and non-economic instruments to control diffuse pollution. However it should be apparent that in practice no one policy can be deemed most cost-effective without considering the site-specific physical and informational characteristics of the polluted control area.

Given the information asymmetry between regulators and farmers and the presence of transaction costs the possibility of an *ex ante* efficient solution is virtually impossible (Smith and Tomasi 1995) and the choice of a second-best solution is quite clearly an empirical issue (Dunn and Shortle 1988; Helfand and House 1995).

Some commentators (Constanza et al. 1995) have argued that there are three necessary criteria to judge an integrated modelling framework, i.e. *realism, precision* and *generality*. The first two considerations relate to simulating system behaviour in a qualitatively and quantitatively precise way respectively, while the generality refers to the range of system behaviour. More importantly they argue that no single model can maximise all these goals concurrently as there are fundamental trade-offs involved in modelling these criteria. Thus such trade-off in objectives must be clearly defined as they determine the modelling approach. Most empirical works trade-off realism with generality and use their results to investigate the overall magnitude and direction of change.

Some empirical work relies on a production function approach, or a bio-physical simulation model integrated with an economic mathematical programming framework (linear and nonlinear), some studies have relied on a *multi-objective* programming framework to consider the interaction between multiple pollutants, while others have used *chance constrained* or probabilistic programming to deal with the stochastic nature of pollutant transport. Fewer researchers have used dynamic programming. Studies also differ in their particular emphasis, i.e. some are concerned with spatial aspects (zoning), while others with the interaction between different inputs, or surface/groundwater interaction, other with transaction costs, equity considerations, uncertainty, temporal issues etc.

Following the distinction in the literature, the empirical studies reviewed below have been divided into *disaggregate* (site specific farm or catchment level models) and *aggregate* (regional or national level) models. Additionally it is important to recognise the insight offered by empirical works not utilising economic instruments as control measures. Such studies can be of considerable interest to economists if they use novel/complex modelling techniques or integrate realistic biophysical considerations and interdependency between polluting decisions/outcomes. It is for this reason that some empirical studies not utilising economic instruments have been discussed at the end of each section. Certain differentiating or interesting aspects of some empirical studies in this chapter are indicated by italicised headings.

4.2 Disaggregate Models

Disaggregate models involve the use of a biophysical simulation model[55] linked with a mathematical programming model representing the economic incentives and policy scenarios. Such models are most commonly found in the literature and can be divided on the basis of their scale into farm (field) level and catchment (or watershed) level models.

It is interesting to note that earlier models of NPS pollution were predominantly at the catchment scale. Studies which have not used economic instruments have been discussed if they have used a novel methodology or if their results offer insights of use to regulators, e.g. using a mixed integer programming framework with stochastic specification (Halstead et al. 1991) or the use of stochastic environmental risk modelling (Teague et al. 1995).

[55] A model of natural processes pertaining to animal husbandry and crop growth.

4.3 Farm level Disaggregate Models

A relatively simple investigation of the effects of alternative crop rotations on nitrogen leaching to groundwater from corn cultivation in Iowa (US) under the US Food Security Act (1985) was undertaken (Huang and Uri 1992). The 7 crop rotations considered, involved combinations of corn, soybean and meadow plantation. The other measures they considered included a) elimination of current price support programme, b) corn sales tax, c) nitrogen tax, d) nitrogen quota. They estimated the farmer's compliance cost to reduce 'excess nitrogen' i.e. the difference between the total amount of nitrogen applied from all sources to one acre of cropland and the amount of nitrogen removed by corps at the end of the growing season. Their results show that crop rotation patterns which limit nitrogen fertiliser use impose the lowest cost to farming while the sales tax is most costly.

A very simple study of nitrate pollution from a single crop in Finland adapted a nitrogen loss function from the literature and compared a nitrogen tax, nitrogen quota and product price reduction (Miettinen 1993). It concluded that nitrogen quotas were the least cost farm level control policy.

Another simple one dimension hydrological model of nitrate leaching for a single crop in Norway investigated the impact of nitrogen fertiliser taxation (Botterweg et al. 1994). They estimate that a 200 – 350% increase in nitrogen input price is required to bring about a significant reduction in leaching. However a lower input tax may confer indirect benefits due to input substitution in favour of manure which is less polluting in principle and confers other ecological benefits such as improving soil structure and biomass content.

Multiple Production Function Specifications

One study (Sumelius 1994a) researched the ability of economic instruments to control NPS nitrogen pollution in Finland. He examined four policy alternatives a) a nitrogen tax which approximately doubled the price of nitrogen, b) a 50% output tax, c) a mixed instrument combing fertiliser and output taxation, d) a per ha fertiliser quota. He estimated the crop production function for 2 crops and derived a leaching function from the literature. The results established 1) *the overall efficiency of input taxation* 2) the product and fertiliser tax combination is more efficient than the product tax alone, but less than the nitrogen tax or quota, 3) yield is not that responsive to fertiliser price increases or product price decreases, 4) yields are reduced most by N quotas.

More interestingly he examined the relative ranking of instruments based on different production function specifications i.e. quadratic polynomial ($y = ax + bx^2 + cx^3$), a square root polynomial ($y = a\sqrt{x} + bx + \ldots$), and Mitscherlich ($y = y_{\max}[1 - \exp(-a - bx)]$) – where x is the nitrogen input. He discovered that under a Mitscherlich specification nitrogen taxation was the most efficient, however under a quadratic specification it was ranked below the nitrogen quota. However the remaining policy options were not affected by production function specification.

Additionally in deriving the marginal abatement cost (MAC) curves he illustrates how the most cost-effective instruments are those which are directed at reducing high initial N leakages. Unlike Kampas (Kampas and White 2002) he believes a change in cropping area may impact more on leakages than a change in fertiliser intensity.

Mixed Integer Linear Programming Framework

Another study (Swinton and Clark 1994) evaluated policies to reduce nitrate leaching using a mixed integer linear programming (MILP) model. They modelled a 500 ha farm growing five crops at 1993 prices on sandy loam soils under two crop rotations in Michigan. They assumed that 40% of the applied mineral nitrogen fertiliser is leached as nitrate, leaching occurs in the year of fertiliser application and that biologically fixed nitrogen does not leach at all. They examined the following five policies: a) obligatory use of the US integrated farm management programme b) a nitrogen fertiliser tax, c) a 'quasi deficiency' payment for specified crop rotations, d) a tax credit (subsidy) for nitrogen predicted to be fixed biologically by rotational crops, e) elimination of current price support (deficiency payments) programme. They concluded that the last option was the most efficient 'second-best' policy to contain nitrogen leaching from both a private and government perspective (financial only).

On a cautionary note they emphasise that the results are limited by the specification of price/cost relationships, technical parameters and that they ignore general equilibrium or 'second-round' effects on future relative prices. They advocate better crop and soil fertility management, i.e. introducing cover crops, which can potentially reduce soil erosion and leaching by '*improving nitrogen uptake during the crop season and immobilizing soil nitrogen during the off season*' (Swinton and Clark 1994). A summary of farm level NPS control empirical studies is presented in table 4.1.

4.3.1 Farm Studies without Economic Instruments

Mixed Integer and Stochastic

A mixed integer programming model of a representative dairy farm in Virginia (US) was used to determine the impact of stochastic constraints on nitrate groundwater loadings (Halstead et al. 1991). Their study linked an economic sub-model of 3 crops and manure management with 20 year CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems) simulation model (Kinsel and Walter 1980) which utilises slope, tillage practices, and temperature besides soil type and rainfall. CREAMS cannot model movement to or within the groundwater system – it only modelled nutrients to the end of the root zone.

The authors advocate the use of stochastic constraint specification because nitrate contamination is subject to stochastic 'pulses' or variability both *within and between years*, due to the weather and management processes. Using probabilistic constraints they analysed a 20% and 40% reduction at the 80% and 90% confidence level. Not surprisingly their results show that a stochastic specification can *increase* regulation costs above a deterministic one (the cost increasing with constraint tightening), and that this extra cost is an 'insurance' against chance violation of the loading constraint. They propose that policy makers should consider a 'two tiered' standard comprising of the *average* annual nitrate standard and a probabilistic bound on its violation.

Table 4.1: Farm Level NPS Control Empirical Studies

Study	Modelling approach	Heterogeneity		Crop rotation s	Instruments Considered						Efficiency verdict
		crops	soils		(1) Nitrogen Tax	(2) Nitrogen Quotas	(3) Reduction in Product Price	(4) Removing Price Support	(5) Subsidy for adopting BMP's	(6) Nitrogen tax + price support removal	
Mietten (1993)	Production function				√	√	√				(2)
Huang and Uri (1992)	Production function	2	?	7	√	√	√	√			(2)
Huang and Lantin (1993)	Production function	2	?	7	√	√	√	√			(2)
Sumelius (1994)	Production function	2		many	√	√	√			√	(1) + (2)
Swinton and Clark (1994)	Mixed Integer Linear Programming	5	1	2	√			√	√		(4)

MOTAD Risk Analysis

A study of stochastic environmental risk from *nitrate and pesticide* pollution modelled a farm in Oklahoma (US) (Teague et al. 1995). They used a target MOTAD programme for producers who wished to maximise net expected returns, but were concerned about maintaining environmental risk indices (both *surface and ground water*) for pesticides and nitrates below critical targets. Stochastic measures of environmental risk can produce different prescriptions from deterministic ones, as the probability distribution of nitrate percolation is often skewed, i.e. the expected value of nitrate loading may not be high but the probability of a large nitrate emission might be.

The research used EPIC-PST[56] to generate nitrate and pesticide emissions, as well as crop production using actual 20 year weather data. Some 5000 production activities differing in terms of nutrient use, irrigation scheduling and pesticide strategy (a minimum of 6 herbicides and 8 insecticides) were included for each of the 3 crops grown on 2 soil types. Three target percentage reductions of 25%, 50% and 75% were imposed to determine the trade-off between net returns and environmental risk by imposing restrictions on the pesticide environmental index, nitrate environmental index and both indices together.

The permitted farmer responses included 1) re-allocating land to production on heavier soils, 2) crop substitution, 3) reducing per acre nitrogen applications, and 4) increased use of fallow rotations. Results showed that although indices involve value judgements and assumptions, overall expected income is more sensitive to nitrate loading restrictions than to pesticide loading restrictions. They also confirm the difference between stochastic and deterministic model specification.

Overall it should be noted that the production function approach, prevalent in the early literature, assumes that the intensive margin effects of policies are more important than the extensive margin ones. However in reality there is considerable

[56] A mixture of EPIC (Erosion Productivity Impact Calculator) and GLEAMS (Groundwater Loading Effects of Agricultural Management Systems)

input substitution between land and fertiliser application (Bouzaher and Shorgen 1995). Secondly, two problems with farm level studies are apparent a) how does one define a representative farm, and b) how can the effects of a policy at farm level be aggregated or extended beyond farm boundaries? For this and other reasons researchers have turned to catchment-scale studies.

4.4 Catchment level Disaggregate Models

One of the earliest works to empirically test the least cost property of economic instruments at the catchment level modelled the San Joaquin valley in California[57] (Horner 1975). The study involved comparing the cost of water treatment for nitrates with that of a tax on nitrogen fertiliser. A nitrate loss function was econometrically established from data from 38 tile drains in the valley. This was linked with a multi-period linear programming model which maximised the present value of future returns to the management of the land. The study concluded that the cost of treatment exceeded the cost of control. Similar results were established by latter work (Braden et al. 1994) which concluded that within certain risk parameters nitrate pollution from agriculture is more cost-effectively prevented than treated, i.e. removed from drinking water.

A catchment level model of the River Gipping (England) compared the cost-effectiveness of two types of a marketable pollution permit system with a regulatory standard to control nitrate leaching (Hartley 1986). The nitrate leaching model was combined with a linear programming model. The first permit system was an overall catchment reduction in nitrate fertiliser usage, while the second was based on specific reductions in the nitrate concentration of the catchment area's water supply. Not surprisingly both permit systems were more cost-effective than the standard, however under the second MPP system a standard was more efficient for really stringent compliance levels. Other early studies of NPS pollution at the watershed/catchment level modelled sediment loses (Lovejoy and Lee 1985) and sediments with nutrients (Kramer et al. 1984) using a basin segmentation approach.

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[57] Subsequent studies on diffuse nitrogen pollution modelled this valley as well (Helfand 1995; Larson et al. 1996)

Spatial Variability

Miltz et.al. (Miltz et al. 1988) question the superiority of a uniform discharge tax over a uniform discharge standard in the presence of spatial variability. They modelled diffuse sediment loadings into a watershed in Illinois (US) using a simulation model called SEDEC which estimates farm budgets, soil loss, *sediment delivery* and *spatial considerations* of land management.

Although an emission tax encourages firms with low MACs to reduce discharges, if the firm's contribution to ambient pollution at the receptor site is low i.e. they have low transfer coefficients (i.e. if there is a positive correlation between MAC and transfer coefficients) then the discharge tax is *not* always superior. This analysis does not consider transaction costs. Secondly, the greater the transfer coefficient[58] variance relative to the MAC variance the more likely the occurrence of a 'cross-over'. This study acknowledges and relies on previous work by Nichols (Nichols 1984) to explain its results.

Land Use Permits

A model of ground water nitrogen pollution in the Cambridge chalk (England) combined a hydro-geological model with a linear programming model (Pan and Hodge 1994). They assume there are 12 land use activities and four nitrogen fertilising regimes in the catchment, and that farmers follow ADAS farm management practices. They utilise average annual percolation rates and soil conditions thus the *variation in leaching within and between years is not accounted for*. The objective of the study was to find a cost-effective means to meet the EU limit on drinking water nitrate concentration. The limit was converted into a per hectare annual nitrogen load by simplifying and assuming various physical factors.

The three policy options examined were a) a fertiliser input tax, and b) a tax on nitrate leaching and c) land use permits. The last option is a simple permit system, assuming the total catchment leaching corresponding to the standard and the nitrate

[58] The transfer coefficient is the sediment delivery ratio in this study.

leaching per hectare from different land uses (based on soil type and land use) can be estimated. Such a system requires that the regulator monitor land use and permit holdings.

Their results indicate that land use permits are more efficient that fertiliser taxation but not as efficient as pigouvian taxation. This ranking was achieved after tax payments are returned to farmers as transfer payments. Overall they are critical of fertiliser taxation because: a) inelastic demand for fertiliser; and b) there may not be a direct link between nitrogen application and leaching, e.g. legumes. The potential transaction cost savings of land use permits is examined in later works (Hodge 1997).

Irrigation

The following 5 empirical studies investigate the control of irrigation water restriction on nitrogen leaching.

An early study of economic instruments to regulate nitrate and leaching and soil erosion examined controls on both irrigation water and nitrogen application (Pfeiffer and Whittlesey 1978). The authors incorporated an annual sediment and nitrate leaching loss function in a linear programming model of the Yakima Basin (US). Their results indicate that irrigation taxes are superior to nitrogen taxes and that a combination of both is better than separately taxing each alone.

Johnson et.al. (Johnson et al. 1991) investigated the on-farm costs of ground water nitrogen pollution control in Oregon (US), where the climate is semi arid and irrigation is required. Their model consisted of a) a plant simulation model (CERES) to estimate yield, soil water and nitrogen balances, b) a two state dynamic optimising model for scheduling irrigation and fertilising decisions for each crop, and c) a *linear* programming model to account for the rotational constraints on a representative farm. In the model four main crops[59] were grown on two soil types and it was

[59] They were unable to run CERES for alfalfa due to lack of data. Therefore although yield estimates were included in the analysis it was assumed there is no nitrate leaching from alfalfa.

assumed that farmers knew with certainty the current fertility and moisture states. They primarily tested three policies: a 100% input tax, a pigouvian or leachate tax, and optimal timing and application of water and nitrogen inputs. They found leachate taxation to be more efficient that input taxation but doubted its practical applicability due to monitoring and enforcement costs. They found the elasticity of demand for nitrogen fertiliser to be low, i.e. high input taxes are required to substantially reduce nitrate leaching.

They concluded that optimal *timing and application* of fertiliser and irrigation water, would result in less total nitrate leaching, a slight increase in yields, and greater profit for the crops and irrigation technology modelled. Under their optimal solution the number of fertiliser applications was 2- 3 times higher under current practices.

In testing the effect of site-specific information i.e. soil heterogeneity, a study compared the effect of uniform and targeted input taxation to reduce nitrate leaching from lettuce production on two soil types in the Salinas Valley (US) (Helfand and House 1995). One soil type is less porous, requires less irrigation water and nitrogen fertiliser. Given that farmers are able to distinguish the two soil types the more porous one leaches more nitrogen not just because of its physical characteristics but also because of farmer's differential nitrogen applications rates. They used the Erosion Productivity Impact Calculator (EPIC) to estimate nitrate effluent and crop production functions and incorporated them into a linear mathematical programme written in GAMS. Assuming an equal distribution of head lettuce (main crop) fields for each soil type and a regulatory objective of 20% reduction in estimated nitrate runoff per acre, they examined the following policies, a) tax both inputs, uniformly across soil types, b) a uniform reduction in levels of input use, c) tax either water or nitrogen uniformly across soil types, d) limit either water or nitrogen use uniformly across soil types. These were compared relative to separate input taxes for both inputs and soil types – the theoretical optimal, yet practically infeasible solution. Their results concluded that 1) *taxing irrigation water is clearly more efficient tht taxing nitrogen aninput*, 2) the efficiency difference between taxing water, both inputs or even separate taxes for each was not significant. Thus they conclude the efficiency gain of targeted spatial taxation is limited, and that uniform imperfect instruments may not always impose higher social costs. As the model did not include transaction costs they point out that it is not possible to rank the practical efficiency of instruments and that as more heterogeneity is included in the analysis the social cost of uniform instruments is likely to increase.

Helfand reports virtually the same study in another paper but emphasises the near equity in economic efficiency between the water and emission tax (Helfand 1995). Similar results were reported in yet another paper (Larson et al. 1996). However it must be noted NPS nitrogen pollution need not be most efficiently controlled elsewhere by restricting or taxing irrigation water alone. Leaching is an extremely complicated, climate-dependent process, and these studies ignore the variability in weather between years, e.g. the occurrence of a random high rainfall event.

Another study utilising EPIC considers nitrate pollution controls in the presence of irrigation in a semi arid Monegros–Flumen region of Spain (Murillo et al. 2001).There is no explicit reference to the target reduction in nitrate leaching, presumably because EU limits are implicitly assumed. The model accounts for 6 crops irrigated with surface water. Although the authors state that relevant soils, tillage and farm operations were included in the analysis, details are not mentioned. The study compares three production function specifications a) polynomial (allows substitution between inputs), b) von Liebig (defines maximum yield, such that further input use does not increase yield), and c) Mitscherlich-Baule (displays both input substitution and maximum yield property). However due to the complexity and convergence problem associated with the Mitscherlich-Baule functional form the polynomial was used.

Three scenarios are examined a) increase in water prices b) increase in nitrogen prices c) a standard restriction on nitrogen use. A water price increase reduces percolation and nitrate leaching substantially (50 – 60%) but at high cost to farmers. Additionally the authors identify the problem of salt accumulation on the surface of certain soils if water use is reduced. An increase in the price of nitrogen reduces percolation as much as the first policy but nitrogen leaching falls by only 26 – 36%. The cost to farmers is lower than under the water price increase. Lastly a standard on nitrogen use proves to be the most cost-effective in reducing nitrogen leaching (62%) and percolation (50%) with the lowest reduction in farmer's net margin. However their analysis does *not include transaction costs*, and they recognise the difficultly of enforcing a nitrogen standard. They recommend that monitoring and enforcement responsibility should be assigned at the district level through water quality measurements of irrigation district return flows. How the district authorities can differentiate between farmers in this way was not mentioned.

The authors also analysed the introduction of better irrigation technology i.e. sprinkle irrigation. They found sprinkle irrigation to be extremely efficient as water and nitrogen utilisation does not fall, net margin increases, but percolation and nitrate leaching is reduced by 37 and 61% respectively. They estimate that the cost of introducing the new irrigation technology is substantially prohibitive and government incentives should be provided.

Another study of irrigated crops in Spain (Cordoba) examined the trade-off between foregone compliance cost and environmental benefits by using a multi-objective programming framework (Zerki and Herruzo 1994). The multi-objectives of the problem included a) maximising gross margin, b) minimising nitrogen leachate, c) minimising nitrogen fertiliser applications and d) minimising drainage water applications. They used the NTRM simulation model to estimate the drain flow and nitrate emissions for different combinations of management practices. Their results show that the voluntary adoption of best management practices yield a reduction of losses ranging between 6 - 21%, whereas considerably high nitrogen taxation is required to achieve the same level of abatement. The interesting policy option of taxing drainage water proved more efficient than the nitrogen input tax. Although easier to monitor than leaching, in practice tracing the drainage water to the source field/farmer is not possible over a large spread of land.

Targeted (Spatial Variation)

A study of *groundwater* nitrate pollution in Oregon (US) assessed the importance of spatial variance in physical parameters in the design of a regulatory tax policy (Fleming and Adams 1997). They modelled five major crops grown on four soil groups, where all representative farms within a soil zone are treated identical. The dynamic modelling was complicated and comprised of three sub-models: a) an economic model which maximises profits subject to an exogenous regulatory policy; b) a one dimensional sub-model which simulates the movement and transformation of nitrates from irrigated land; and c) a groundwater transport model which tracks loading and movement of nitrates within the aquifer.

The research examined the effect of a spatially differentiated tax based on soil zoning, i.e. each zone is taxed differently but farmers within each zone are taxed the same. Their results concluded that the efficiency gains from introducing spatially differentiated taxation based on soil types are negligible and likely to disappear on considering the associated transaction costs.

The hydrological model sets it apart from the Helfand studies where the objective was to control the quantity of nitrate leached into the root zone - this does not necessarily translate into actual groundwater concentration due to dilution, delay and groundwater flow. Another difference is that this study did not consider any irrigation water controls. This study is also very interesting because it highlights the complexities of the real world. The model calculated that once a tax is introduced it takes 5 to 10 years for soil water nitrate concentrations to return to a steady state, but groundwater nitrate concentrations require approximately 118 years to achieve stability[60]!

Multiple Pollutants

There has also been research which attempts to integrate policy controls of *nitrogen, phosphorus and sediment* (soil erosion) (Vatn et al. 1997). The authors contend that the degree of interaction (interdisciplinary) and resolution of their model

[60] Where stability is defined as the state where the annual change in concentration across all cells is less than 0.3 parts per million.

ECECMOD, is comparable in sophistication with CEEPES (Bouzaher and Shorgen 1995) and NERC/ESRC Land Use Programme or NELUP (NELUP 2000). ECECMOD is comprehensive, comprises of various non-linear sub-models, and accounts for spatial factors such as topography and farm position, besides agronomic practices such as tillage options, manure handling and spring-time management. It is realistic because its decision procedure is consistent with how information about the growing season is sequentially revealed to the farmer in practice; thus decisions are based on expected (average year) considerations. The study area is South Eastern Norway and the regulatory objective is that N and P must be reduced by 50%. The policies analysed are a) a 100% tax on N input, b) a 50% arable land requirement on catch crops/grass cover, and c) a per ha subsidy for spring tillage/ no tillage.

The results show that the input tax is the least costly in terms of the per hectare and per kg reduction in N leached; whereas catch crops are most costly, however the introduction of catch crops do control P and soil erosion considerably. In combining catch crop requirement and input tax they found the combined effect was less than the sum of their separate effects. Whereas the tax (reducing N losses) and spring tillage (effecting soil erosion) combination does work without reducing their separate effects.

Cost Benefit Analysis

A study of an catchment in Ontario (Canada) compared the benefits and costs of nitrate pollution reduction (Giraldez and Fox 1995). The benefit of water quality improvement was derived from estimates of toxicological and epidemiological data from the literature. CREAMS was used to simulate surface runoff, root zone leaching and thus enable estimation of the on-farm cost of improving water quality. They conclude that a 55% nitrogen input tax is sufficient to induce optimal abatement. In addition they estimate that the on-farm cost of nitrate abatement and the cost of using bottled water is less than the off-farm benefits of nitrate abatement.

Transaction Costs and Stochastic

A study of the Kennet catchment in England examined the affect of transaction or administrative costs on the policy ranking of instruments to control NPS nitrogen pollution (Kampas and White 2002). Farm heterogeneity is modelled through 3 different soil types which were determined using GIS mapping. Two biophysical simulation models (NCYCLE and SUNDIAL) along with a simple hydrological model (TOPCAT) were used to generate production and emission functions (it was assumed that emissions were log normal in distribution) which were incorporated in a stochastic non-linear programming model written in GAMS. They used certain simplifying assumptions regarding the drainage volume carrying the nitrogen load to the subsoil, and in the conversion of the EU water nitrate concentration standard (mg/liter) into an equivalent per hectare load (kg). The policies analysed included emission permits and taxes, uniform and targeted input quotas, nitrogen tax, land tax, and setaside restrictions. They used some questionable 'best available proxies' for administrative costs based on the per hectare cost estimate of current agri-environmental schemes. Their instrument ranking is based on social costs, i.e. abatement or resource cost (tax payments are considered transfer payments and excluded) plus administrative or transaction costs.

Their results show a) price control policies outperform quantity control policies for both emission and input restrictions, b) they report modest improvements with the use of targeted input quotas, c) if considering only abatement costs emission taxation is the most efficient and land retirement is the most costly option and d) in terms of social costs nitrogen input taxation is the most cost-effective control option.

A noteworthy finding of the study was that the policy ranking varied depending on the required regulatory stringency. For example setaside is an extremely costly control at high regulatory reliability (i.e. greater required likelihood of achieving standard) however at lower reliability is ranking improves. Their results show that the abatement cost of different control regulations do not exhibit 'cross-over' (unlike the previous findings based on spatial differences (Miltz et al. 1988)), but the administrative cost frontiers do intercept.

Spatial Zones

Utilising Geographical Information Systems (GIS) a British study (Moxey and White 1994) divided the Tyne catchment (England) into *land classes* (based on soil type, climate and gradient) and two hydrological *zones* (or two sub catchment boundaries). Five year average nitrogen response and emission coefficients for different production activities were estimated using EPIC and incorporated into a linear programming model. It was assumed that nitrate emissions were only diluted by the flow of discharged leachate (i.e. there was no ground water contribution and that river flow is comprised of mainly surface water) and carried to the receptor point in the lower zone by a delivery ratio of one (for upland to lowland). Their regulator objective was to meet the EC Nitrate Directive standard, and they examined three different quota systems a) tradable emission quotas, b) tradable nitrogen quotas and c) targeted nitrogen quotas for each land class, with no trade between classes

They found that a nitrate emission quota was the most efficient, followed by the targeted nitrogen quota outperforming the uniform quota by a margin which increases as the abatement requirement increases. However no transaction costs estimates were included in the analysis. Research has also highlighted the potential benefit of using GIS for accurate integrated modelling at the regional/national level by linking the microparameter distribution model and GIS (Opaluch and Segerson 1991). Overall there is considerable interest in the use of GIS (Cook and Norman 1996; Moxey 1996).

Groundwater Aquifer

A study of the US southern high plains overlaying the Ogallala aquifer (Wu et al. 1995) compared the use of the following policies to control ground water nitrogen leaching: a) a per-acre nitrogen restriction; b) nitrogen input tax; c) tax on irrigation water; and d) incentives to use modern irrigation technology. As the semi-arid study area has diverse physical features it was mapped using GIS and split into two sub-regions. Nitrogen runoff and leaching from 4 crops (plus summer fallow) grown on 4 representative soils, using 4 different irrigation technologies, phosphorus, nitrogen and pesticides was determined by running EPIC-PST by inputing 20 years actual weather data.

The above mentioned four policies were ranked both in terms of changes in farm income and social welfare. Results show that incentives to farmers to adopt irrigation technology outperformed all other policies overall from both society's and farmer's perspective. From society's perspective nitrogen input taxation is preferred over nitrogen input quotas which farmers prefer because they have lower compliance costs. Consistent with other empirical studies they found the demand for nitrogen was very inelastic.

The paper admits that policies which reduce expected nitrogen losses might not reduce 'spikes' in emissions in the event of high rainfall. However they argue that groundwater acts as a buffer and these spikes are not transferred to the consumer, thus 'mean nitrogen runoff and leaching are useful indicators of surface and groundwater contamination potential'. This is true of nitrate leaching to groundwater but not surface water runoff (Shortle et al. 1998; Shortle and Horan 2001). A summary of catchment level NPS control empirical studies is presented in table 4.2.

Table 4.2: Catchment Level NPS Control Empirical Studies

Study	Modelling approach		Model details		Instruments considered										Efficiency Verdict
	Mathematical Programming	Biophysical Simulation model	Irrigation Included	G.I.S.	(1) Emission tax	(2) Emission quotas	(3) Nitrogen Taxation	(4) Nitrogen Quotas	(5) Targeted Nitrogen quotas	(6) Irrigation water tax.	(7) Combination of (3) and (6)	(8) Arable to grassland	(9) Water Treatment	(10) Subsidy for adopting BMP's	
Homer 1975	LP	Estimated N Loss function					√						√		(3)
Pfeifer and Whittlesey 1978	LP	Estimated N Loss function	√				√			√	√				(7)
Hartley 1986	LP	Leeching model from the literature				√		√							(2)
Miltz et al., 1988	LP	SEDEC			√	√									(2)
Johnson et al., 1991	LP	CERES	√		√		√								(1)
Pan and Hodge 1994	LP	Hydrological model from the literature					√		√						(1)
Giraldez and Fox 1995	LP	CREAMS	√				√						√		(1)
Zerki and Herruzo 1994	MLP	NTRM	√				√			√				√	(6)
Moxey and White, 1994	LP	EPIC		√			√	√	√						(2)
Helfand and House 1995	PF	EPIC	√				√			√	√				(6)
Wu et al.,1995	LP	EPIC- PST	√	√			√	√	√	√				√	(10)
Vatn et al., 1997	NLP	ECECMOD		√			√					√			(4)
Murillo, et al., 2001	NLP	EPIC	√	√											(6)
Kampas and White, 2002	NLP	NCYCLE, SUNDIAL TOPCAT		√		√	√	√	√			√			(3)

4.4.1 Catchment Studies without Economic Instruments

Nutrients, Pesticides, Sediments and Stochastic

A seminal work not utilising environmental instruments comprehensively modelled a watershed using a stochastic programming framework to generate probability distributions of nitrogen, phosphorus, pesticides and sediments pollution (both loads and concentration) to surface and groundwater from agriculture (Milon 1987). Milon was interested in the surface/ground water interaction resulting from control policies. The example sited is that of conservation tillage which:

'*substitutes increased herbicide and pesticide usage for conventional cultivation practices. By reducing water runoff, however, conservation tillage increases subsurface water recharge and increased likelihood of groundwater contamination*' (Milon 1987).

This study is also unique because it was the first study to relate field losses to flow rates and receiving surface and groundwater quality (concentration). One earlier work of the Great Lakes (US) used a mass balance model to calculate surface water phosphorus concentrations (Chapra et al. 1983). Milon used PRZM, STREAM and AT123D models to estimate pesticide and sediment concentrations whereas HSPF was used to determine phosphorus and nitrates concentrations entering receiving water bodies. These relationships were incorporated in a stochastic optimising framework developed some 24 years earlier (Charnes and Cooper 1963).

A basin in Ohio was modelled as 4 soil types grown on 3 crops for *29 years* actual weather, utilising 3 tillage practices, various combinations of 8 pesticides, herbicides, nutrient levels and crops rotations. Various reliability levels of reductions from the baseline scenario were examined. Although no economic instrument was explicitly modelled the study concludes a) that in determining the offset ratio in a point/nonpoint trading policy a consistent reliability criteria should be applied across all sources, b) the control of one environmental externality can have detrimental impacts on the level of another (there is a trade-off).

Spatial Modelling

A study of the dairy farms in the Lower Susquehanna watershed (US) used *spatial information* (slope, proximity to water, soil type etc.)to reduce the costs of controlling NPS nitrate pollution (Carpentier et al. 1998). Based on certain assumptions they estimated the 10 year average information, contracting and enforcement costs (the sum of which make up transaction costs) of implementing a performance based standard which aims to reduce N delivery by 40%. 237 farms are modelled in SUSFARM (an LP programme written in GAMS) which distinguishes 36 crop rotations comprising of 9 crops, 4 tillage technologies, and the option of contour stripping. Nitrate emissions were generated in EPIC and a mixed integer programming model ALLOCATI was used to minimise the catchment cost of achieving the target reduction. Where catchment cost are the sum of compliance costs (approximate shadow prices of nitrate reduction on each farm) and transaction costs.

Farmers had the option of reducing runoff by strip-cropping and manure management for both uniform and targeted policy controls. Targeted policies utilising spatial information (GIS and surveys etc.) reduced compliance costs by nearly 80% when compared to uniform policies. With targeted policies transaction costs were also reduced as fewer farms required contracting and enforcement of the performance standard. Overall total compliance costs were 75% lower with targeted polices.

Sediment Movement Control

A novel study of sediment control in Illinois (US) watershed analysed the option of containing the movement of emissions (Braden et al. 1989). The study used SEDEC, a programme which simulates profit, erosion and transport of sediments (based on parameters such as slope, soil type, topography, position of drainage network etc.) under various management practice alternatives. They modelled 3 crops, 4 crop rotations, six tillage practices, three structural measures to prevent sedimentation and set a 50% regulatory reduction target. Surprisingly crop rotations were the most efficient option and not structural practices - this was attributed to the gentle slopes. They also found strategic or targeted control measure were a lot more cost-effective.

Such containment is not relevant to the control of nitrates as current technology does not economically permit restricting the movement of nitrate emissions once generated. This study is mentioned because it illustrates the effectiveness of targeted and control policies and the importance and site specific characteristics/features.

Integrated Control of Externalities

Another study compared policies to control sediments from *soil erosion, nitrate* and *atrazine* (herbicide) leaching in a Iowa watershed. They disregard the piece-meal approach targeting particular water pollutants separately and stress the complex interaction between pollutants, e.g. increased chemical use with conservation tillage which may lead to reduced water quality. They address the fact that not only do economic and environmental objectives conflict, but that there is conflict between environmental objectives themselves (Lakshminarayan et al. 1995). They investigate these significant trade-offs using a multi criteria decision–making approach based on multi-attribute utility theory. To account for site-specific variability at the large scale without having to perform thousands of site-specific simulations, they use a procedure known as metamodelling[61] which a simple response function fitted to the 'biogeophysical' outputs from calibrated mathematical simulation models. They model a 25 million acre cropland watershed[62] in Iowa (US) by using a linear programme (not probabilistic or dynamic) which models 18 crop rotations, 2 conservation, and 4 tillage practices.

Their broad aims were to examine soil quality protection, groundwater quality protection and a combination of both. They concluded there was a significant trade-off between economic and environmental goals, and even among environmental goals themselves. Policies targeting soil quality tend to adversely impact

[61] A metamodel is a regression model explaining the input-output relationship of a process model. It is essentially a *'statistical approach which abstract away from detailed regional analysis by approximating outcomes of a complex process model through statistically validated parametric forms'* (Bouzaher et al. 1993).

[62] A drainage basin or catchment is the area of land that drains water, sediment, dissolved materials, and biota to a common outlet at some point along a stream channel. If the area is large (hundreds of square miles) it is a drainage basin, if it is small (acres to square miles) it is a catchment. *Watershed* is technically defined as the topographical divide (drainage divide) that separates catchments. In the United States, a 'watershed' means a catchment and 'drainage divide' means a watershed (UCCE 2002).

groundwater quality and vice versa, and thus a compromise solution can only be reached at high social cost. These results of significance to regulatory authorities and underlie the importance of integrating pollution control policies across different pollutants. For a discussion and empirical study of the use of metamodelling in investigating the comprise between elements which determine water quality (sediment, nutrient, chemicals, bio-toxins) see previous work by the author (Bouzaher et al. 1993). The need to integrate policies to control different (mutilple) agricultural pollutants has been emphasised by other commentators too (Connor et al. 1995).

Another empirical work which investigated the trade-off between groundwater pollution and soil erosion modelled an irrigated catchment in Oregon (US) (Connor et al. 1995). Their empirical work was based on integrating the relationships derived from CERES (nitrogen leaching model) and FUSED (sediment loss mode) into a multi-objective programming model. Their results highlight the need to integrate and coordinate the control of both sediment loss and nitrate groundwater pollution.

Equity

A very interesting study of atrazine (pesticide) and sediment (soil erosion) pollution in a watershed in Illinois (US) (primarily attributed to increased use of conservation tillage in a bid to control soil erosion!), utilises an indicator of *equity* to ensure that the economic burden of pollution control is distributed evenly (based on an exogenously specified equity indicator) between representative farms with differing environmental impacts (due to spatial variability and soil differences) and hence abatement costs (Önal et al. 1998). The equity indicator is based on absolute deviations from a uniform distribution of economic outcomes (each farm's income losses), and captures the difference in economic control costs of farms when subject to environmental regulation.

Using GIS the catchment was divided into sub-basins and then seven representative farms, growing 3 crops on 6 different soil types (based on soil, slope, landscape complexes) utilising 3 different tillage practices. A chance constrained or stochastic specification is used to limit atrizine runoff in a linear model written in GAMS. Their results show that the income distribution constraint does reduce economic efficiency, as expected, but the efficiency loss is less than 10% of the total costs of environmental regulation. The paper concludes with an ex post comparison with *actual incentive payments* to farmers in the watershed. The values of these payments were found to be approximately equal to the losses estimated by the model when the burden of regulation is equally shared among farmers. Obviously the result is catchment specific and depends on the stringency of standard, the required reliability, the degree of heterogeneity between farms and abatement cost structures

Overall, although in regulatory terms catchment scale policy ranking is an improvement upon farm level results, most regulators are concerned with implementing and enforcing a national or regional policy. Thus it is important to aggregate or extend the analysis beyond catchment boundaries if possible. This of course involves loss of targeted or site specific information and a move towards more generic policy formulation.

4.5 Aggregate Models

This section will discuss empirical aggregate models which are larger in geographical scale than the catchment and encompass regions or national levels. A large study of a 48,500 sq mile Central Plains region over laying the Ogallala Formation (US) was modelled as five sub regions to test the potential gains of targeted policies to control stochastic nitrogen runoff and percolation (Mapp et al. 1994). The model is comprehensive and accounts of alternative tillage practices, 4 irrigation technologies at 6 levels, pesticide applications, 3 crops and 4 soil types. They employed EPIC-PST and MODFLOW (groundwater flow) to simulate production and the movement of nitrates using 20 years of actual weather data. They policies analysed included: a) an overall quota on nitrogen application; b) a per hectare nitrogen quota; c) a targeted nitrogen quota on coarser/permeable soils prone to leaching; and d) a targeted quantity restrictions on specific irrigation regimes (furrow). The results are presented as *cumulative distribution functions* of annual nitrate runoff and percolation losses. Their results show that per-acre restrictions are likely to be more efficient than total nitrogen restrictions in reducing runoff and percolation at all reliability levels. Secondly, targeting soil types did not produce the expected efficiency gain due to the distribution of soils (heavier soils are dominant, coarser ones are few) and because producers had already moved intensive production off soils prone to emissions; therefore targeting production systems is better. Finally, the income loss under targeted policies is less than under broad ones because fewer total acres are targeted. They do not however consider the income distributional effects between individual farmers.

Another way of aggregating site-specific pollution problem involves the use of microparameter distribution models (Wu and Segerson 1995a). Microparameter models use joint probability distributions of micro or firm/farm level parameters such as production and pollution functions (Opaluch and Segerson 1991). Wu and Segerson studied the control of groundwater nitrate pollution in Wisconsin (US) for which they derived acreage elasticities with respect to site-specific characteristics and policies to analyse the extensive margin impact of commodity programmes on nitrate groundwater pollution. Three polices were examined a) reduced target corn price, b) increase in the APR rate for corn, c) nitrogen input tax. The APR scheme is an annual voluntary land retirement scheme where participating farmers setaside a proportion of their arable land to qualify for benefits such as loans and deficiency payments. An interesting outcome of the approach is that it is not possible to calculate the social welfare impact of policies as acreage changes (extensive margin) cannot be immediately translated into changes in consumer or producer welfare. Their main finding was in favour of the policy increasing the APR payment for corn.

In an extremely large simulation modelling exercise (Bernardo et al. 1993a; Bernardo et al. 1993b) researchers investigated relative efficiency of polices to control nitrate pollution to groundwater in the High Plains Aquifer. The terrestrial area of the aquifer region modelled was over 125,000 km square and spread over 5 US states. They sub-divided the region into five agro-regions which they defined as agricultural regions exhibiting similar ecological, political boundaries[63] and farming characteristics (soils, climate, production practices etc.). Their simulation modelling is comprised of a) a crop growth/pesticide and nitrate percolation and runof transport model (EPIC-PST), b) a regional linear mathematical programming model which simulates decision making over a 20 year horizon and c) a groundwater flow model (MODFLOW) to determine future groundwater pumping scenarios. They investigated the effect of: 1) a per unit acre restriction which limits nitrogen application by one third; 2) a total upper bound on farm nitrogen application; and 3) the effective banning of certain pesticides. Their results concluded that although a total quota was the most effective policy to control runoff losses, a per unit acre limit was the most effective tool to control nitrate percolation. Overall however, a total restriction on nitrogen pollution had a lower compliance cost to producers than a per unit-area restriction.

This is a very interesting result at it implies that efficient policies to control nitrate pollution differ depending on the type of nitrate pollution i.e. runoff to surface water or percolation to groundwater.

Some commentators have listed problems associated with aggregating data at regional levels (Wu and Segerson 1995b), whereas others (Constanza et al. 1995) have argued the inherent difficulties of aggregating ecological systems due to their non-linearities and irreducible complexities.

Table 4.3 is a summary of regional NPS control Empirical studies.

[63] Although it is not necessary that agro-region boundaries correspond to political boundaries it is often the most practical division as economic data are reported at this level.

Table 4.3: Regional Level NPS Control Empirical Studies

Study	Modelling approach		Model Details		Instruments considered					
	Mathematical programming	Biophysical simulation model		GIS	Total Nitrogen Quotas	Per-acre Nitrogen Quotas	Nitrogen Tax	Output Price Reduction	Modification of Price support Policies	Efficiency verdict
Bernado *et al.*, (1993)	Recursive LP	EPIC-PST MODFLOW		√	√	√				Inconclusive
Mapp *et al.*, (1994)	Recursive LP	EPIC-PST MODFLOW		√	√	√				Total Nitrogen Quotas
Wu and Sergerson (1995a)	Microparameter Distribution Model			√			√	√	√	n/a

4.6 Studies Integrating Water Quality and Quantity

Studies which do not consider the issue of water quality and quantity interaction at the catchment level; rather at the aquifer, field, or valley 'portion' levels are discussed in this last section.

A non-economic study involving field trials and laboratory analysis of potato production in the US (North-Central region) investigated the impact of different irrigation schemes (sprinkler and drip), irrigation triggers, various sources of N and their timings on nitrate leaching to groundwater (Waddell et al. 2000). The study reports that 40% deficit irrigation, five N applications splits, drip irrigation, reduced N leaching significantly and has a minimal impact on potato tuber yield and quality. Thus it can be concluded that *irrigation and nitrogen management has a definite impact on the control of nitrogen leaching to groundwater under potato cultivation.*

Another study of cotton production in a 'portion' of the San Joaquin Valley (US) investigated the impact of 1) increased water prices, 2) adoption of water conserving technologies (four different irrigation technologies), and 3) the imposition of a pollution tax, in terms of yields, water use, profitability and the quantity of drainage effluent (Caswell et al. 1990). The research concludes that the adoption of modern irrigation technology (sprinkle or drip irrigation) would reduce the quantity of contaminated drainage water by a) reducing pollution per acre, and b) reducing the quantity of wate applied per acre. Secondly, the introduction of a pollution tax should reduce water use and pollution by a) reduce water use and pollution generation on farms, b) encouraging adoption of more efficient and less polluting irrigation technology, and c) providing incentives to retire low quality lands.

An interesting study of regulating both water quantity and quality in California (US) involved a 20 year dynamic model of 3 producers of cotton and alfalfa reliant on a limited supply of surface water irrigation and common property *aquifer* (Dinar and Xepapadeas 1998). In the unregulated case each producer acts myopically and the percolation of irrigation water pollutes the aquifer. The results show that there is no convergence to a steady state; without monitoring quality, deterioration will result in water over exploitation. Two regulatory scenarios were examined, 1) involving a central monitoring case where taxes and quota are applied to surface water uniformly across all producers, and 2) individual monitoring (using observation wells) of water withdrawls and pollution by each producer allowing a flat tax on pollution volume and a pollution volume quota. The regulator's objective function is to maximise total regional income; tax receipts are assumed to be returned to the region for investment in regional water improvement activities.

Their results show *a surface-water tax appears to be more efficient than surface water quota in achieving both better environmental standards and higher regional income*. They also argue that by incurring costly monitoring, the regulator can use pollution taxes which ensure a steady state of better quality and higher level of water in the aquifer. Additionally individual monitoring was found to be superior to central monitoring. However these results should be viewed with caution because they are based on certain assumptions. A similar study of quantity and quality management of groundwater in Greece derived the optimal taxes under cooperative and non-cooperative solutions (Xepapadeas 1996).

Yet another study of San Joaquin Valley, California (US) examined the impact of water markets on water conservation, economic efficiency and the environmental consequences of drainage reduction using a micro-level production model i.e. in a single crop and single field setting (Dinar and Letey 1991). They conclude that water markets enable farmers to both invest in more efficient irrigation technology and pay for the safe disposal of drainage produced on their fields. It must be remembered that the model was very simplistic and does not capture any substitution between inputs, or farm heterogeneity – to name a few limitations.

A later study extended the above analysis by examining the impacts of changes in a) irrigation practices, b) water prices and allotments, and c) the introduction of water markets on five crops grown on 70,000 hectares of the San Joaquin Valley (US). The regulatory target was to achieve a 30% drainage reduction to limit the impact of selenium. They found that water markets were less cost-effective than input or effluent based taxes to control pollution. This was attributed to the different incentives, i.e. a) water markets created a general incentive to reduce water use while the taxes motivated conservation of only that water applied in excess of plant requirements, and b) unlike water markets tax instruments created a direct incentive to reduce consumption of all instruments contributing to effluent production. However, interestingly, they claim that the transaction costs of implementing water markets are lower than those of input taxation (observation and monitoring costs) therefore potentially making controlling diffuse pollution through water markets a more efficient means to control pollution. They argue that water markets provide a positive environmental impact.

4.7 Discussion

The problem with absolute comparisons of empirical studies is that no two studies assume the same parameters, or model the same degree of complexity, management practices, etc. Thus economists generally rely on relative instrument/policy ranking.

Empirical works differ on the basis of 1) their efficiency criteria, i.e. whether they are concerned with farmer costs or social costs, 2) transaction costs, the inclusion of which changes policy ranking, 3) stochastic specification, i.e. whether standards target the mean alone or the mean and variation in pollution, 4) the inclusion of spatial factors such as slope and topography, 5) degree of farm/catchment heterogeneity, i.e. number of crops and soil types (the use of GIS and its resolution) 6) whether tillage and irrigation technology has been accounted for, 7) the prevalent management practice of farmers in terms of type, timing and application method of artificial fertilisers and their substitutes, 8) whether input substitution is accounted for, 9) climatic differences or weather patterns, 10) the number of actual years weather the simulation is run for, 11) the crop rotations considered, 12) the regulatory standard and its stringency (required degree of attainment), 13) whether the objective is to control surface (runoff) or groundwater (leaching) pollution, 14) the catchment scale and type, i.e. surface water or groundwater catchment, 15) whether pollution reduction is required at the root zone or actual aquifer groundwater, 16) whether the regulatory objective is in terms of load (kg) or concentration (mg/l), 17) the biophysical model used, its assumptions and calibration to site-specific conditions (sophistication), 18) whether the modelling framework allows for discrete decision making e.g. choice of technology or adopting a subsidized management scheme, and 19) whether interaction with other water pollutants is considered, for example the use of conservation tillage may decrease sedimentation but will probably increase herbicide use. The differences do not end here and the above is not a definitive list.

Some of these factors determine how close the model baseline is to the reality. Other considerations include behavioural assumptions regarding farmer's risk preference, strategic behaviour, and utility maximisation by maximising profits alone. In reality the political feasibility of a control measure is probably as important a consideration as efficiency (Hahn 1990; Keohane et al. 1997; Aidt 1998). In modelling and parameter setting there is a degree of subjectivity, which must be kept in mind. All of the above makes empirically modelling pollution control an extremely complicated procedure and as any comparison of empirical studies must consider the above, it is apparent why comparative analysis and policy ranking on the basis of one study does not resulted in one 'true policy ranking'. However, once aware of these caveats it is safe to consider the general conclusions of empirical studies; some of which are presented below.

Firstly it must be remembered that these studies are limited to a partial equilibrium analysis utilising consumer and producer surplus changes as measures of welfare. In reality environmental pollution control policies are likely to change prices (depending on the level they were implemented) and incomes, of farmers those linked to farming, thus a more relevant measure of social cost would be compensating or equivalent variation (Hazilla and Kopp 1990). A general equilibrium analysis of the welfare effects of implementing the Clean Air Act and the Clean Water Act in the US found it to be significant (Hazilla and Kopp 1990).

Numerous studies have confirmed the least cost property of taxation (emission or input) (Horner 1975; Pan and Hodge 1994; Giraldez and Fox 1995; Wu et al. 1995;

Kampas and White 2002). However some studies have identified 'cros-overs', i.e. ranges in which taxation is sub optimal in comparison with regulator standards (Nichols 1984; Miltz et al. 1988; Bouzaher et al. 1990).

On the question of which input to regulate there are conflicting results. Four empirical studies of semi-arid agriculture where irrigation water is a necessary input fail to conclude the optimality of taxing one input over the other, although it seems as if a conclusion based on one years weather (Larson et al. 1996) conditions is not as convincing as one based on many years weather.

Although targeted polices are theoretically more efficient that uniform ones (Kolstad 1987) two empirical studies outlined above conclude that the efficiency gain is quite small (Helfand and House 1995; Fleming and Adams 1997). However other studies which compare the welfare gains of targeted polices with estimated transaction costs (Carpentier et al. 1998) estimate tremendous long term cost-effectiveness. In fact the overall majority favour the efficiency of the targeted policies (Fox et al. 1995; Shortle et al. 1998), but whether practical enforcement and administrative cost considerations allow it is another matter (Moxey and White 1994; Dion et al. 1998). Another interesting study demonstrated the inefficiency of applying uniform policies, and showed that this inefficiency was dependent upon the extent of spatial variability in costs relative to benefits and the correlation between them (Fox et al. 1995; Babcock et al. 1997).

As argued in chapter 2, price support distortions provide false incentives to farmers to over produce by both intensification and extensification of agricultural production which invariably increases nitrate loading to the environment. In fact one empircial study concluded that phasing out price support would be the most cost-effective control of nitrate pollution (Swinton and Clark 1994). However due to the sensitive income effect of this policy on farmers, and due to the sensitivity of resolving environmental and agriculural considerations such controls are not vociferously advocated (Hrubovcak et al. 1990; Abler and Shortle 1992).

Better management practices (crop rotations, timing, application of fertilisers etc.) and technology (irrigation, tillage) have repeatedly been advocated as efficient policy options (Braden et al. 1989; Johnson et al. 1991; Huang and Uri 1992; Mapp et al. 1994; Swinton and Clark 1994; Wu et al. 1995; Vatn et al. 1997; Murillo et al. 2001). Thus implying there should be more incentives, like cost sharing programmes and education, to encourage adoption of better management and technology.

Trade-off between policy objectives has been established. This trade-off exists not only between different types of water pollutants but between different water bodies receiving the same pollutant. It has been demonstrated that the control of one externality can have a detrimental impact on the level of another agricultural externality (Milon 1987; Lakshminarayan et al. 1995), similarly the optimal policy to control nitrate runoff may be different from the one which contains nitrate leaching (Murillo et al. 2001).

Only one study explicitly accounts for transaction costs[64] (Kampas and White 2002), the fact that so many studies have ignored transaction cost estimates is a major failing for economists, and without doubt adds to the general 'impracticality' associated with economic solutions to environmental pollution.

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[64] Another research (Carpentier et al. 1998) also estimated transaction costs but did not use economic instruments.

Chapter 5

Internalising Agricultural Surface Water Externalities Theoretically

5.1 Introduction

The aim of this chapter is to theoretically model the internalisation of two surface water externalities in an agricultural catchment. The two agricultural externalities considered are *non-point source nitrogen pollution* and *reduced river flows from surface water irrigation*. The previous two chapters detailed theoretical and empirical developments in the efficient control of non-point source nitrogen pollution from agriculture[65]. To date, numerous aspects of diffuse nitrogen pollution as a *negative* production externality have been examined (Shortle and Dunn 1986; Segerson 1988; Xepapadeas 1992a; Braden and Segerson 1993). This chapter will consider the dual nature of surface water diffuse nitrogen pollution both as a *positive and negative externality* in a catchment where a regulator wishes to enforce a minimum river flow and water quality standard.

The first part will introduce the concept of '*complimentary interaction*' between controls which target different agricultural externalities, i.e. surface water NPS nitrate pollution (quality) and low river flows (quantity). The next section will detail positive production, while the final section theoretically integrates complimentary interaction and the dual nature of NPS nitrate pollution as a positive and negative externality.

5.2 Complimentary Water Quantity/Quality Interaction

Given a function of crop yield in terms of *both nitrogen and water* it is possible to demonstrate the contribution of both the application of water and nitrogen inputs in the generation of NPS nitrate pollution. Thus both inputs should be investigated to determine whether there are efficiency gains from the regulation of both rather than one in the control of NPS nitrate pollution.

Consider a catchment comprised of one farm (one decision maker) producing a single crop, for simplicity. The regulatory objective is to maximise catchment profit subject to *two environmental standards*, one pertaining to nitrogen emissions, E^* (water quality) and the other to surface water extraction for irrigation, X_2^* (water quantity). Where x_1 and x_2 refer to nitrogen and water inputs respectively. The water extraction standards can be thought of as the maximum extractable water which will ensure compliance with a *minimum river flow* restriction. The need for minimum river flows arise because surface water extraction by irrigators reduces natural flows to levels which are damaging to catchment ecology and other users, i.e. recreational usage, fishing etc.

Furthermore assume the production and emission functions capture the variability in the catchment weather; i.e. both are based on expected weather patterns. The emission function and regulatory standard are stated in terms of load or mass (kg) of nitrogen being carried to the receiving water body. Although a rather crude assumption, it can be excused on the grounds that it remains illustrative while simplifying notation considerably. The regulatory objective is to maximise catchment profits subject to the environmental standards E^* and X_2^*, the emission and water extraction standards respectively:

$$\max_{x_1 x_2} \pi = pf(x_1 x_2) - wx_1 - hx_2 \qquad \text{(EQ 5.1)}$$

Subject to:

$e \leq E^*$ Emission Standard (EQ 5.2)

$x_2 \leq X_2^*$ Water Extraction Standard (EQ 5.3)

Where, $e = s(x_1, x_2)$

[65] There is an abundance of theoretical reviews of NPS control instruments in the economic literature (Dosi and Tomasi 1994b; Weersink et al. 1998; Ribaudo et al. 1999; Shortle and Horan 2001)

Setting up the Lagrange (with λ and z as Lagrange multipliers):

$$L = pf(x_1 x_2) - wx_1 - hx_2 - \lambda\left(E^* - s(x_1, x_2)\right) - z\left(X_2^* - x_2\right) \tag{EQ 5.4}$$

F.O.C

$$\frac{\partial L}{\partial e} = \lambda \tag{EQ 5.5}$$

$$\frac{\partial L}{\partial x_1} = pf'(x_1^*) - w + \lambda s'(x_1^*) = 0 \tag{EQ 5.6}$$

$$\frac{\partial L}{\partial x_2} = pf'(x_2^*) - h + \lambda s'(x_2^*) + z = 0 \tag{EQ 5.7}$$

The optimal solution reveals that to ensure the standards are not exceeded the following two taxes must be levied (either based on estimated or actual emission):

Emission tax $\lambda = \dfrac{w - pf'(x_1^*)}{s'(x_1^*)}$ (EQ 5.8)

Surface water extraction tax $z = h - pf'(x_2^*) - \lambda s'(x_2^*)$ (EQ 5.9)

Since λ, i.e. the emission tax (EQ 5.8) enters the surface water extraction tax equation (EQ 5.9) it is evident that the optimal emission tax determines the optimal irrigation tax level. This relationship can be called 'complimentary interaction', because the control of one input compliments the control of the other. It is safe to assume that $s'(x_2^*)$ is positive because greater water application on soils rich in nitrate fertiliser will probably increase emissions by washing it out. However it is possible, given dry conditions, that at initial levels of crop growth the availability of water allows for more uptake of nitrogen and reduces emissions i.e. the value of $s'(x_2^*)$ may be negative. Whether the value of $\lambda s'(x_2^*)$ is significant enough to make a significant impact on z[66], is an empirical issue and will undoubtedly be catchment and climate specific.

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66 The equivalent input tax is $\lambda \cdot s'(x_1^*)$, whether this is larger or smaller than the emission tax depends on whether $s'(x_1^*)$ is < or > 1. However this does not necessarily indicate which instrument shall have the greater compliance cost, that depends on the input demand elasticity of nitrogen.

The result is intuitively obvious. The regulatory instrument reducing nitrogen usage, also affects the level of the other corrective instrument targeting water extraction. One explanation is that within the range of nitrogen input normally applied, there exists a complimentary relationship between the crop uptake of water and nitrogen, thus lowering the use of one necessitates reducing the consumption of the other.

Alternatively it may be argued that restricting irrigation water usage reduces crop yield and quality (thus market price and profitability) - thereby reducing the economic incentives to use nitrogen. For example on the East Coast of Scotland, where rainfall is plentiful, irrigation water contributes to tuber quality or better 'finish', i.e. less lines and reduced incidence of scab etc. The potato crop may be divided into scabbed and scab free, the former being associated with inadequate or 'restricted' irrigation, while the latter with optimal or higher irrigation. The difference is reflected in the market price of potatoes as the 'optimally irrigated' crop has a higher scab free proportion thereby fetching a significantly higher price. By restricting irrigation the regulatory authority will lower the profitability margin per hectare and shift land allocation from optimal to restricted irrigation thereby reducing incentives to apply as much of the other input, nitrogen.

To the regulator who seeks to control both externalities by ensuring least cost compliance with environmental standards the result of complimentary interaction between instruments is significant. The fact that taxation of one externality determines the optimal control of another implies the two problems should be considered jointly; doing so should reduce the compliance cost.

The literature (Helfand 1995; Larson et al. 1996) has theoretically and empirically modelled the efficiency gain of regulating water as opposed to nitrogen input directly in a bid to curb NPS nitrate pollution - a review of other empirical studies investigating the efficiency of which input should be regulated has been presented in chapter 4. It should be noted that what differentiates this research from previous work is that previous literature has considered the control of NPS nitrate pollution by limiting irrigation water application or creating water markets – no specific minimum acceptable river flow (MAF) restrictions have been considered. Whereas the analysis presented in this chapter (and onwards) considers the efficiency gain in NPS nitrate pollution control when MAF restrictions are enforced.

5.3 Positive Production Externalities in a First-best World

This section concerns internalising the potential positive production externalities associated with diffuse nitrogen pollution in a first-best world without irrigation. In an ideal world the regulator has perfect information (Braden and Segerson 1993) regarding the contribution of emission to environmental damage (ecological and human). The regulatory objective is to maximise catchment profits (a proxy for utility) while internalising environmental externalities. Assume the entire catchment can be spatially split up (Moxey and White 1994) into two zones A (upstream) and B (downstream). Each zone comprises a single decision making farmer. Assume that the downstream activity requires irrigation of arable land. Upstream nitrogen emissions (from artificial nitrogen or farmyard manure) enter the river and form a portion of the irrigation water applied downstream. Thus upstream emissions act as positive externality to downstream farmers. The value of this contribution can vary from negligible to marginal depending upon the area of upstream land, the intensity of arable agriculture and the reliance on irrigation downstream.

Assume instantaneous interaction i.e. no time lags in the movement or mixing of emissions. Furthermore both farms produce different output dependent on their unique known production functions (f_1, f_2) while the amount of their output is insignificant in proportion to the market, thereby making them price takers.

Upstream Emissions (Zone A)

$$e_1 = E_1(x_1) \qquad \text{(EQ 5.10)}$$

$$a_1 = A_1(e_1) \qquad \text{(EQ 5.11)}$$

Upstream emissions depend on the amount of nitrogen fertiliser applied x_1. Emissions are stated as the load or mass of nitrogen carried to the receiving water body. The volume of water which carries emissions and the actual river flow rate determines the overall ambient concentration of nitrogen upstream a_1. The environmental damage d_1 to society (anglers, cross-country walkers etc.) in zone A from fertiliser applications arises if

$$d_1 = \begin{cases} d_1(a_1) & if \quad a_1 > a^* \\ 0 & if \quad a_1 \leq a^* \end{cases} \tag{EQ 5.12}$$

where a^* is the equilibrium concentration representing the assimilative capacity of the environment.

Downstream Emissions (Zone B)

Downstream nitrogen emissions depend on the actual nitrogen applied by the farmer x_2 (artificial or farmyard manure) and the nitrogen content (e_I) of the applied irrigation water from the river u_2, and h is the associated irrigation water extraction and application cost per unit u_2. The nitrogen concentration of irrigation water applied downstream is determined by a_1, or farming activities upstream.

$$e_I = g(u_2, a_1) \tag{EQ 5.13}$$

$$e_2 = E_2(x_2, e_I) \tag{EQ 5.14}$$

Assuming no collusion/bargaining between upstream and downstream farmers implies the farmer in zone B does not know the exact level of e_I. Thus the ambient nitrogen concentration of river water is a) a positive production externality to farming downstream if e_I contributes to production i.e. $x_2 + e_I \leq x_2^{\max}$, where $x_2^{\max}$ is the input level corresponding to the maximum physical production (not economic) i.e. $\dfrac{\partial f_2}{\partial x_2} = 0$ or b) a negative production externality if $x_2 + e_I > x_2^{\max}$ and $\dfrac{\partial f_2}{\partial x_2} < 0$, i.e. stage III of production where the diminishing or toxic effect or both of nitrogen application occur. Therefore whether e_I, which is provided free of charge, is a positive or negative production externality depends on $x_2 - x_2^{\max}$ or the downstream farmer's knowledge of e_I. Thus the ambient nitrogen concentration downstream is a_2 and the environmental damage d_2.

$$a_2 = A_2(e_1, e_2) \tag{EQ 5.15}$$

$$d_2 = \begin{cases} d_2(a_2) & if \quad a_2 > a^* \\ 0 & if \quad a_2 \leq a^* \end{cases} \tag{EQ 5.16}$$

Assuming that the farmers are risk neutral, profit maximising and have no influence on the prices of input or outputs, a simplified[67] social net benefit function which maximises the difference between the expected benefits of polluting activities and the resulting environmental damages is:

Regulator's welfare problem (First Best World):

$$\underset{x_1, x_2, u_2}{Max} \; (p_1 f_1(x_1) - w_1 x_1) + \left(p_2 f_2(x_2, e_1^I, u_2) - w_2 x_2 - h u_2\right) - d_1(a_1) - d_2(a_2)$$

(EQ 5.17)

F.O.C.

$$p_1 f_1^{'}(x_1^*) + p_2 \frac{\partial f_2}{\partial e_I} e_I^{\,'}(x_1^*) - \left(\frac{\partial d_1}{\partial a_1} \frac{\partial a_1}{\partial e_1} e_1^{'}\left(x_1^*\right) + \frac{\partial d_2}{\partial a_2} \frac{\partial a_2}{\partial e_2} \frac{\partial e_2}{\partial e_I} \frac{\partial e_I}{\partial a_1} \frac{\partial a_1}{\partial e_1} e_1^{'}\left(x_1^*\right) \right) = w_1$$

(EQ 5.18)

$$p_2 \frac{\partial f_2(x_2^*)}{\partial x_2} - \frac{\partial d_2}{\partial a_2} \frac{\partial a_2}{\partial e_2} e_2^{'}(x_2^*, e_I) = w_2 \tag{EQ 5.19}$$

$$p_2 f_2^{\,'}\left(u_2^*\right) - \frac{\partial d_2}{\partial a_2} \frac{\partial a_2}{\partial e_2} \frac{\partial e_2}{\partial e_I} e_I^{\,'}\left(u_2^*, a_1\right) = h \tag{EQ 5.20}$$

Upstream emissions contribute to environmental degradation (negative impact) while also fertilising crops downstream (positive impact). In setting the corrective pigouvian tax the regulator must account for both opposing impacts. In choosing the optimal tax levels the regulator equates each input's net marginal benefit with the marginal social cost. As the regulator has perfect information he is able to enforce the social optimum by levying two spatially differentiated emission taxes equal to the aggregate marginal external 'effects' of each farmer (Xepapadeas 1997).

Upstream Tax

$$T_1 = \frac{\partial d_1}{\partial a_1}\frac{\partial a_1}{\partial e_1}e_1'\left(x_1^*\right) + \frac{\partial d_2}{\partial a_2}\frac{\partial a_2}{\partial e_2}\frac{\partial e_2}{\partial e_I}\frac{\partial e_I}{\partial a_1}\frac{\partial a_1}{\partial e_1}e_1'\left(x_1^*\right) - p_2\frac{\partial f_2}{\partial e_I}e_I'(x_1^*) \qquad \text{(EQ 5.21)}$$

Downstream Tax

$$T_2 = \frac{\partial d_2}{\partial a_2}\frac{\partial a_2}{\partial e_2}e_2'(x_2^*, e_I) + \frac{\partial d_2}{\partial a_2}\frac{\partial a_2}{\partial e_2}\frac{\partial e_2}{\partial e_I}e_I'\left(u_2^*, a_1\right) \qquad \text{(EQ 5.22)}$$

It is likely that upstream emission tax might be lower than downstream taxation; because a portion of up stream pollution benefits the farmer downstream. However how they differ depends on catchment specific farming practices, the relative size of zones, ambient nitrate levels before regulation and the particular crops grown in each zone.

5.4 Complimentary Interaction and Positive Externalities: Second-best Solution

This section of the chapter will model complimentary interaction and positive externalities in a second-best world where due to imperfect information on environmental damage estimation regulatory standards are imposed (Beavis and Walker 1983a) . To reiterate, *complimentary interaction* refers to the situation where two inputs contribute to non-point source pollution and where controlling one indirectly reduces the need to control the other.

[67] The is formulation is simplified since it ignores a) other factors of production which may be pollution abating or increasing, and their substitution, b) the entry exit condition on the number of production sites i.e. the acreage of land under cultivation.

Assume the regulator divides the catchment into two zones or farms, upstream and downstream. The regulator's objective is to keep the river nitrate concentration in both zones below E^* while ensuring that river flow in both zones (i.e., r_1, r_2) does not fall below particular standards. Unlike the water quality standard, which is the same for both zones, due to river hydrology complications simplicity requires the quantity standards are stated in terms of upper bounds on allowable water extraction IRR_1^* and IRR_2^* for each zone respectively[68]. Both ensure that that a minimum river flow rate is maintained at any given time of an 'average year'. Assume that the weather pattern realised $\hat{\omega}$ corresponds to an 'average year'

For simplicity assume only one predominant soil type and one irrigated crop, such as potatoes, is grown in the catchment and its market price is fixed at p. It should be noted that land allocation within each zone is an important factor in determining emission concentration, however its inclusion complicates notation considerably. Each farm has its own production function with unique managerial approaches $\hat{m}_1$ and $\hat{m}_2$ which refer specifically to any managerial practices related to nitrogen and water usage only, not any other input. As this is a static representation of a dynamic process it necessitates assuming instant mixing and no lag in the movement of emissions.

Relationships

$r_1 = R_1(x_1^2, \hat{b}_1)$	River flow rate upstream	(EQ 5.22)
$r_2 = R_2(x_2^2, r_1)$	River flow rate downstream	(EQ 5.23)
$c_1 = s_1(x_1^1, x_1^2, \hat{\gamma}_1, \hat{m}_1, \hat{\varpi}, \hat{b}_1)$	Upstream river concentration	(EQ 5.24)
$c_2 = s_2(x_2^1, c_1^I, x_2^2, r_1, \hat{\gamma}_2, \hat{m}_2, \hat{\varpi})$	Downstream river concentration	(EQ 5.25)
$c_1^I = I(c_1)$	Interactive term (concentration)	(EQ 5.26)

[68] In reality they would probably vary by the month/week.

Where x_1^1 and x_2^1 (subscript refers to the upper (1) and lower (2) zones) are the nitrogen inputs (superscript 1), whereas x_1^2 and x_2^2 refer to the irrigation water extraction (superscript 2). Both depend on the area under potato cultivation, plant growth requirements, desired potato tuber finishing, incidence of disease, cost of irrigation etc. c_1^I is the *interactive term* or the nitrate concentration of irrigation water applied to crops downstream as a result of upstream nitrogen fertiliser applications; thus the irrigation water applied downstream x_2^2 has an nitrate concentration of c_1^I.

γ_1 and γ_2 refer to the proportion of total catchment land in each zone and not the land allocation within each zone. The river flow upstream r_1, comprises of base river flow b_1 (given), and all the drainage associated with land in the upstream zone. Upstream flow r_1, also serves as the base river flow to the downstream zone. Essentially upstream decisions affect downstream farming but not visa versa. $\hat{\varpi}$ refers to the weather pattern realised in both zones and any symbol with a hat on it implies it is given and not a decision variable.

An important distinction here is that nitrogen emissions are stated as concentrations, i.e. capturing both the mass of nitrogen and its dilution in the water (both rain and irrigation) carrying it. The regulator's social welfare problem is:

$$\underset{x_1^2, x_1^1, x_2^2, x_2^1}{\text{Max}} \; L = \gamma_1\left(pf_1(x_1^1, x_1^2, \hat{m}_1, \hat{\omega}) - wx_1^1 - hx_1^2\right) + \gamma_2\left(pf_2(x_2^1, x_2^2, c_1^I, \hat{m}_2, \hat{\omega}) - wx_2^1 - hx_2^2\right)$$
$$+ \lambda_1(c_1 - E^*) + z_1(x_1^2 - IRR_1^*) + \lambda_2(c_2 - E^*) + z_2(x_2^2 - IRR_2^*)$$

(EQ 5.27)

The above has been subject to the following constraints, i.e. environmental standards:

$$c_1, c_2 \leq E^* \tag{EQ 5.28}$$

$$x_1^2 \leq IRR_1^* \tag{EQ 5.29}$$

$$x_2^2 \leq IRR_2^* \qquad \text{(EQ 5.30)}$$

The first order conditions are:

$$\frac{\partial L}{\partial x_1^1} = p\left(\gamma_1 \cdot f_1'(x_1^{1^*}) + \gamma_2\left(\frac{\partial f_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c_1}\frac{\partial c_1}{\partial x_1^{1^*}}\right)\right) - w$$

$$+\lambda_1\left(s_1'(x_1^{1^*})\right) + \lambda_2\left(\frac{\partial c_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c^I}\frac{\partial c^I}{\partial x_1^{1^*}}\right) = 0 \qquad \text{(EQ 5.31)}$$

$$\frac{\partial L}{\partial x_2^1} = p\left(\gamma_2 \cdot f_2'(x_2^{1^*})\right) - w + \lambda_2(s_2'(x_2^{1^*})) = 0 \qquad \text{(EQ 5.32)}$$

$$\frac{\partial L}{\partial x_1^2} = p\left(\gamma_1 \cdot f_1'(x_1^{2^*}) + \gamma_2\left(\frac{\partial f_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c_1}\frac{\partial c_1}{x_1^2}\right)\right) - h$$

$$+\lambda_1\left(s_1'(x_1^{2^*})\right) + \lambda_2\left(\frac{\partial c_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c_1}\frac{\partial c_1}{\partial x_1^2}\right) + z_1 = 0 \qquad \text{(EQ 5.33)}$$

$$\frac{\partial L}{\partial x_2^2} = p\left(\gamma_2 \cdot f_2'(x_2^{2^*})\right) - h + \lambda_2(s_2'(x_2^{2^*})) + z_2 = 0 \qquad \text{(EQ 5.34)}$$

Resulting in the following four corrective taxes:

Upstream ambient N tax

$$\lambda_1 = \frac{w - p\left(\gamma_1 \cdot f_1'(x_1^{1^*}) + \gamma_2\left(\frac{\partial f_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c_1}\frac{\partial c_1}{\partial x_1^{1^*}}\right)\right) - \lambda_2\left(\frac{\partial c_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c^I}\frac{\partial c^I}{\partial x_1^{1^*}}\right)}{\left(s_1'(x_1^{1^*})\right)} \qquad \text{(EQ 5.35)}$$

Upstream water extraction tax

$$z_1 = h - p\left(\gamma_1 \cdot f_1^{'}(x_1^{2^*}) + \gamma_2\left(\frac{\partial f_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c_1}\frac{\partial c_1}{x_1^2}\right)\right) - \lambda_1\left(s_1^{'}(x_1^{2^*})\right) - \lambda_2\left(\frac{\partial c_2}{\partial c_1^I}\frac{\partial c_1^I}{\partial c_1}\frac{\partial c_1}{\partial x_1^2}\right)$$

(EQ 5.36)

Downstream ambient N tax

$$\lambda_2 = \frac{w - p\left(\gamma_2 \cdot f_2^{'}(x_2^{1^*})\right)}{(s_2^{'}(x_2^{1^*}))} \qquad \text{(EQ 5.37)}$$

Downstream water extraction tax

$$z_2 = h - p\left(\gamma_2 \cdot f_2^{'}(x_2^{2^*})\right) - \lambda_2(s_2^{'}(x_2^{2^*})) \qquad \text{(EQ 5.38)}$$

As is expected the downstream impact of upstream input use factor into upstream taxation through the inclusion of λ_2 in both upstream ambient and extraction taxes. *Both up and down stream extraction taxes are affected by the ambient taxes in their respective zones*. Thus to ensure cost-effective compliance with the regulatory standards the regulator will have to enforce all four corrective taxes.

The regulatory instruments targeting nitrogen usage also affect the optimal level of instruments controlling water consumption. One explanation is that within the range of nitrogen input normally associated with farming, there exists a relationship between crop uptake of water and nitrogen, thus lowering the use of one necessitates reducing the others application. Alternatively it is possible to reason that restricting irrigation water usage reduces crop profitability and therefore the economic incentive to use the other input.

5.5 Concluding Remarks

Obviously any policy which can differentiate between polluters and their marginal impact is not only more efficient but equitable. Although it is probable that enforcement/transaction costs of implementing a spatially differentiated (targeted) tax will be higher, whether these costs outweigh the efficiency gain over a non-spatially differentiated tax policy is an empirical issue.

This chapter has presented a theoretical outline of two possible efficiency gains a) diffuse N as a positive externality and b) complimentary interaction between NPS nitrate pollution and minimum river flow restrictions. The relative gains in practice would depend on catchment specifics, such as size of catchment (acreage), distribution of zones, extent of irrigation downstream etc. It is possible to extend the analysis by including different soil and crop types; one might even consider the game theoretic implication of upstream and downstream negotiations given the benefit incurred downstream is significant to warrant it.

Similarly a situation might occur whereby the cropping pattern and size of a catchment is such that the regulator increases the number of zones beyond the two modelled here. However disaggregating the effects of different zones and indeed relaxing the assumption of one decision making farmer per zone will result in the problems usually associated with ambient taxation. Incomplete information will probably prevent completely realising the theoretical benefits outlined in this paper. However, any effort which brings us closer to the ideal will definitely confer benefits in the long run.

The empirical component of this research in the following chapters only models the presence of river MAF (minimum acceptable flows) and their impact on NPS nitrate pollution control regulation – not positive production externalities. Positive production externalities were not modelled since it requires complex hydrological and spatial modelling which was not available.

Chapter 6
Catchment Modelling

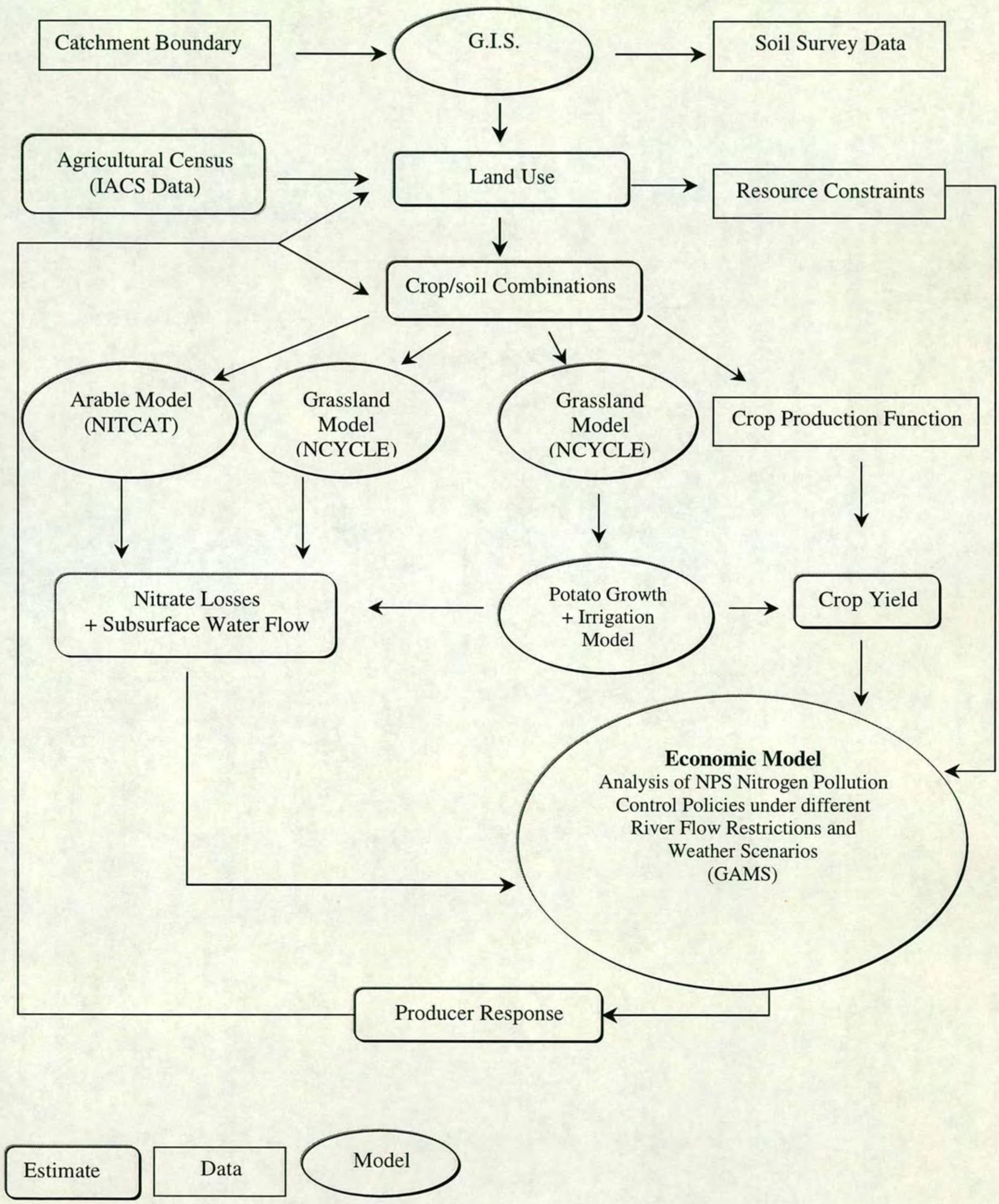
6.1 Introduction

The previous chapter details the theoretical economic benefits of an integrated policy which jointly considers the control of both diffuse nitrate pollution and low river flows. This chapter will outline the framework and assumptions used to model diffuse nitrate pollution as a *negative externality* in the presence of minimum acceptable flow (MAF) restrictions. The West Peffer catchment (East Lothian, Scotland) was modelled because of the prevailing reliance of farming on surface water irrigation and intensive cultivation.

This chapter comprises a) an estimation of catchment soils and land uses, b) a brief description of irrigation modelling c) nitrogen crop production function estimation d) nitrogen leaching function estimation, e) irrigation modelling and potato growth/leaching estimation, f) livestock modelling and, g) an economic model which maximises social welfare in the presence of nitrate regulation and river flow restrictions. A diagrammatic representation of the catchment model and its links is presented in Figure 6.1.

The economic model assumes the catchment is one farm, i.e. one decision maker, whose objective is to maximise returns over nitrogen input use subject to bio-physical resource limitations such as rotational constraints. The arable area of the West Peffer catchment is divided into 3 soil texture categories – *sandy, silty* and *loamy*, on the basis of their texture and drainage properties. It was also assumed that 5 main crops were grown - *winter wheat, spring barley, winter oilseed rape, maincrop potatoes* and *grass* of which only potatoes required irrigation. Grass was subdivided into grazing and cutting (silage). 3 potato crop types were considered based on the level of irrigation water applied i.e. *optimal,*

Figure 6.1: Catchment Model



restricted and un-irrigated. Overall the crops account for 94% of the crops produced on average in the catchment[69].

The West Peffer is a very small low ground catchment located in the East Lothian region of Scotland (see map 6.1). This region comes under the new proposed nitrate vulnerable zones (NVZs) (Ball and MacDonald 2001; Scottish-Executive 2002). In comparison with England and Wales, Scottish catchments experience shorter residence times i.e. water movement from groundwater to rivers is quicker. However this does not rule out dilution of nitrates in groundwater by 'old water', nor does it mean that there is no recharge from purer source of water i.e. non-agricultural land such as forestry etc. There is no known groundwater specific modelling of the catchment to date. The West Peffer catchment is characterised by complex groundwater flows and is 'likely' to be a predominantly surface water catchment (MacDonald 2002). To date there has not been any detailed hydrological modelling of the catchment.

Table 6.1 lists SEPA's measurement of N concentration (mg/l) in the burn at 4 different locations from Jan 1990 – May 2002. This is all the collected data on nitrate concentrations in the catchment. Although the nitrogen concentrations are not particularly high it (i.e. do not exceed the Water Framework Directive 11.3 mg/l nitrogen standard by much) it must be noted that there are only 6 measurements per year of which 3 are in March, May and July year when nitrate losses are comparatively low as nitrate losses tend to coincide with high rainfall events. Additionally given that the catchment is so small and intensively cultivated it is *suspected* that there are complex underground soil processes in the catchment preventing river nitrate levels to rise (MacDonald 2002).

[69] This figure is based on farm level IACS estimates of the proportion of crops grown on farms within the catchment. Of course farm and catchment boundaries do not necessarily coincide.

Table 6.1: Measured West Peffer Nitrogen Concentrations (Jan 1990-May 2002) (mg/l) - Source: SEPA.

Date Measured	Fenton Barns (NT5112880632)	Fenton Barns (NT5038880343)	Congalton Gardens (NT5418580070)	Gauging Station (NT4888081069)
24-Jan-1990	Not Measured	Not Measured	12	8
08-Mar-1990	Not Measured	Not Measured	11.6	12.8
14-Mar-1990	Not Measured	Not Measured	10.8	12.2
07-Jun-1990	Not Measured	Not Measured	4.5	2.5
26-Jun-1990	Not Measured	Not Measured	2.2	1.3
20-Jan-1994	15	15	14	16
24-Mar-1994	14.4	15	13.9	15.3
05-May-1994	10.2	11.4	11.9	11.6
07-Jul-1994	5.34	10.1	1.26	8.29
08-Sep-1994	5.71	7.12	9.75	7.54
27-Oct-1994	4.94	5.64	1.86	6.6
12-Jan-1995	12.3	13.4	12.2	13.7
22-Mar-1995	10.6	11.9	12	11.9
11-May-1995	7.61	7.61	2.9	8.81
20-Jul-1995	5.51	3.56	1.86	2.85
14-Sep-1995	12.2	12.4	Not Measured	12.9
05-Oct-1995	Not Measured	Not Measured	9.86	Not Measured
16-Nov-1995	23.2	23.5	21.8	22.3
25-Jan-1996	13.8	14.8	13.9	14.7
19-Mar-1996	10.9	11.7	12.2	11.7
14-May-1996	8.66	10.5	11.9	9.91
27-Aug-1996	7.99	3.53	10.8	2.65
19-Sep-1996	1.63	5.78	Not Measured	2.05
19-Nov-1996	9.86	9.69	12.2	9.48
29-Jan-1997	**13.7**	**14.8**	**16.2**	**14.5**
27-Mar-1997	**12.2**	**13.6**	**15.4**	**13.5**
20-May-1997	**16.3**	**15.4**	**16.7**	**20.9**
24-Jul-1997	**10**	**11.2**	**12.5**	**11.1**
18-Sep-1997	**6.32**	**7.27**	**11.9**	**7.44**
12-Nov-1997	**8.71**	**9.66**	**11.2**	**9.62**
22-Jan-1998	**21.7**	**23.6**	**19.5**	**22.4**
26-Mar-1998	**13.8**	**14.7**	**16.3**	**13.9**
27-May-1998	**8.8**	**9.66**	**12.8**	**9.29**
28-Jul-1998	**11.6**	**12.5**	**14.7**	**12.2**
23-Sep-1998	**9.64**	**10.2**	**14.1**	**9.68**
18-Nov-1998	**17.1**	**18.4**	**15.2**	**17.2**
27-Jan-1999	17.3	18.7	19.4	17.8
18-Mar-1999	13.4	12.9	16.2	13.1
26-May-1999	9.23	9.68	15.3	8.98
22-Jul-1999	7.05	7.68	13.8	7.64
22-Sep-1999	9.16	9.44	12.9	9.52
16-Nov-1999	7.43	7.92	11.3	7.49

01-Feb-2000	18.1	18.1	17.4	17.8
16-Mar-2000	10.1	10.9	11.9	10.6
25-May-2000	10.6	11.8	13	11
19-Jul-2000	12.5	13	4.73	10.5
05-Sep-2000	7.72	9	11	8.66
25-Oct-2000	9.51	9.96	10.6	10.1
31-Jan-2001	11.4	12.6	12.6	12.4
22-Mar-2001	12.4	Not Measured	13.2	Not Measured
24-May-2001	8.24	9.05	11.7	8.62
18-Jul-2001	7.11	7.8	11.2	7.34
04-Sep-2001	5.39	6.37	10.3	6.37
21-Nov-2001	9.23	9.21	11.61	9.13
30-Jan-2002	14	15.3	14.4	13.2
20-Mar-2002	9.31	9.77	12.2	12
16-May-2002	7.09	7.92	10.8	7.2

(Ordnance Survey map grid references are in parentheses)

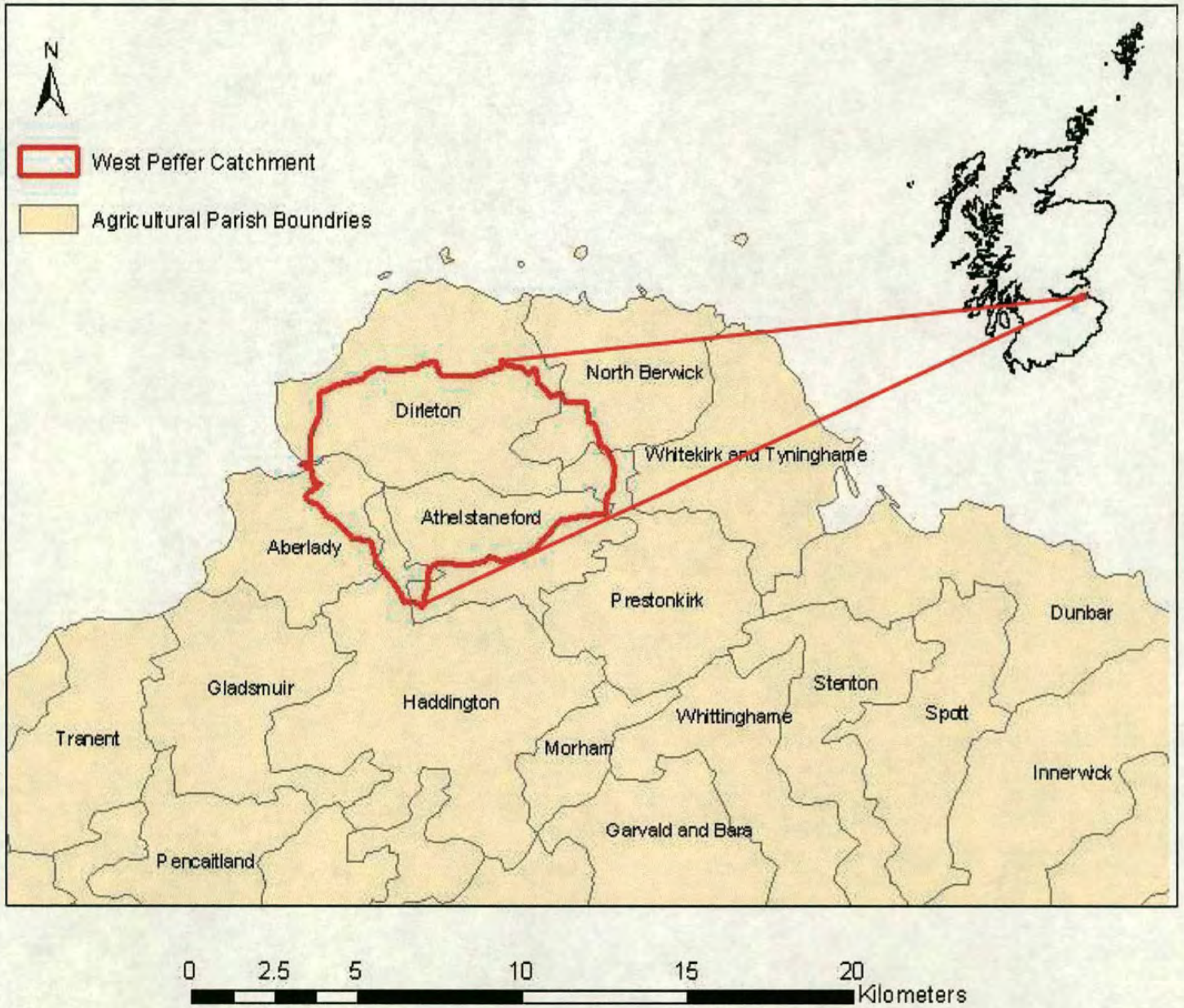
6.2 Catchment Soils

This section describes the spatial distribution of agricultural activities. The MLURI holds a database comprising approximately 12,000 descriptions of soil profiles collected from throughout Scotland in order to characterise soil mapping units. Information such as depth of soil layers and soil texture can be derived from this dataset for all soils in Scotland.

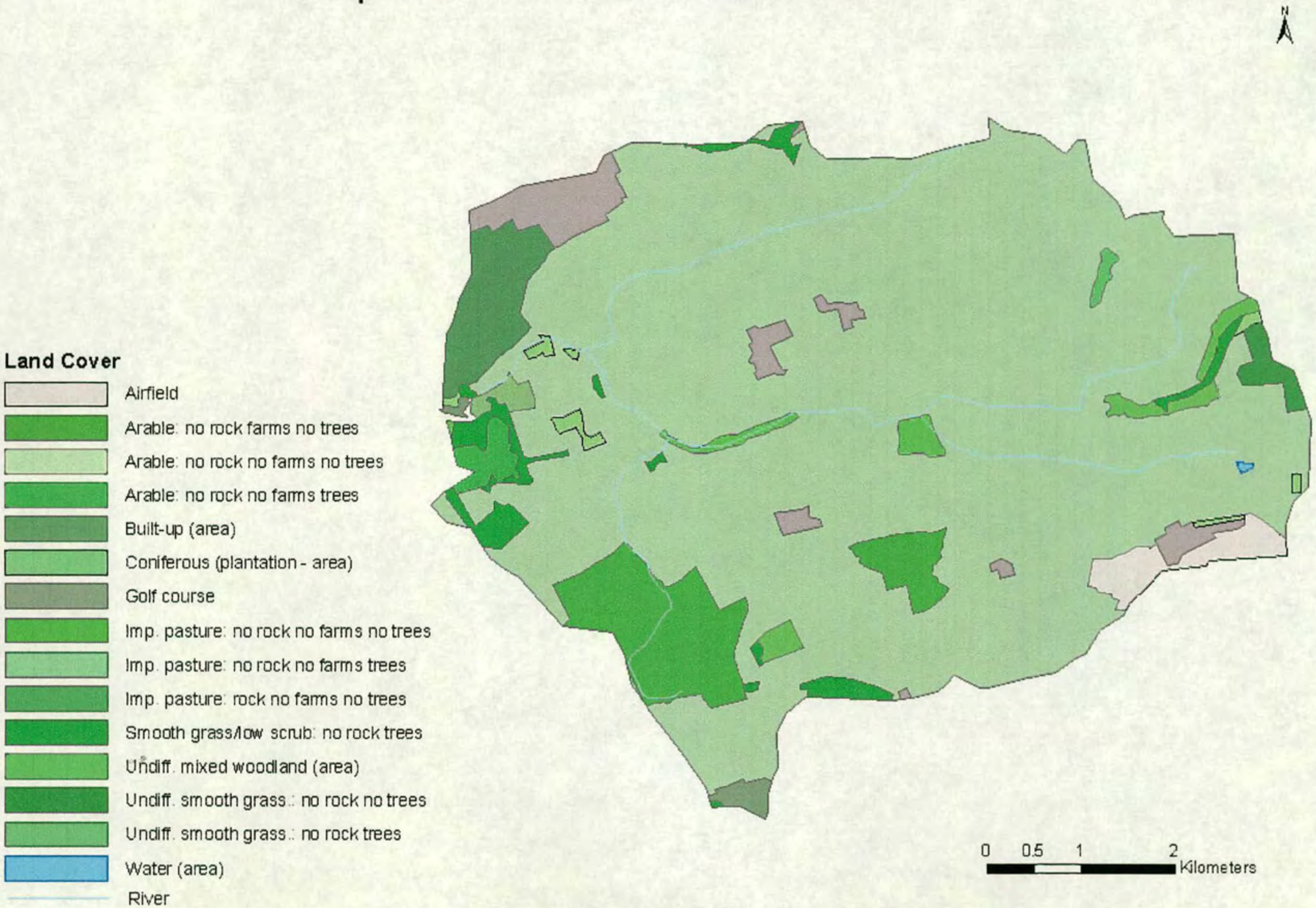
Soil maps of scale 1: 25,000 are available for approximately 50% of Scotland including the West Peffer catchment (Ragg et al. 1966). Many of these maps have been digitised and the digital dataset stored within the ARC/Info geographic information system (GIS). The West Peffer catchment boundary is also held in digital format (ordnance survey 50m digital elevation model) and the soils that lie within the catchment can be easily determined by simple overlay within a GIS. Map 6.3 shows the distribution of different soils in the catchment.

In general, soil map units at the 1:25,000 scale, are considered to be homogenous and to comprise only one soil taxonomic unit (a soil series) with the map unit bearing the name of the taxonomic unit. However, natural variability often means that these map units contain more than one soil taxonomic unit – even though at the time of mapping care was taken to ensure other soil taxonomic types within a map unit had similar physical/chemical properties to those of the dominant soil type. Where this

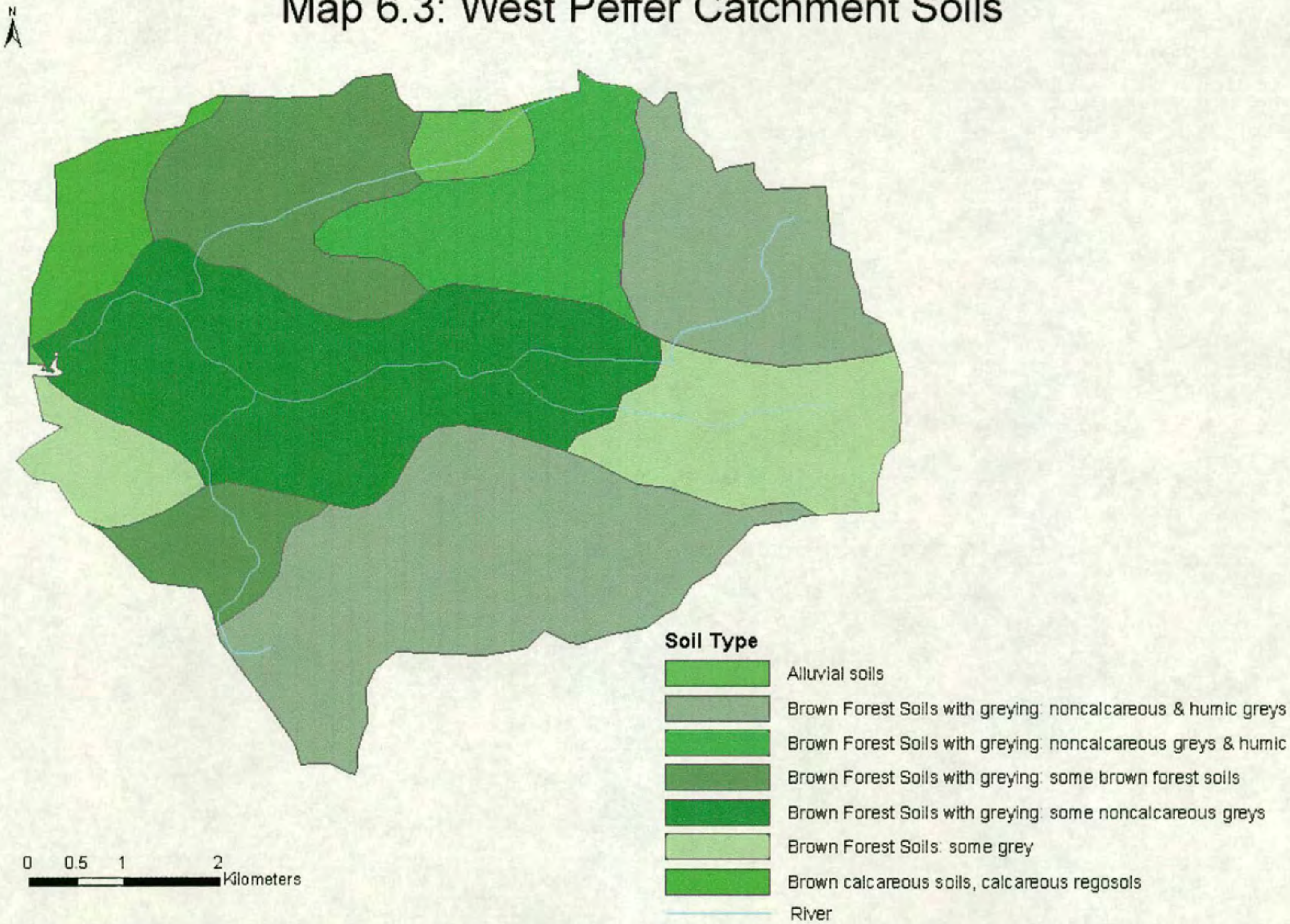
Map 6.1: West Peffer Catchment



Map 6.2: West Peffer Landcover



Map 6.3: West Peffer Catchment Soils



was not possible, the map units were deemed 'soil complexes' and the inclusion of other taxonomic units clearly stated. The soils were grouped on the basis of their drainage and texture properties into *sandy, silty* and *loamy soils*, the results are presented in table 6.2.

6.3 Catchment Land Use Estimation

Normally catchment scale studies of agricultural pollution calibrate their baseline land allocation and livestock numbers based on annual Agricultural Census data. This census data is available at the parish or sometimes grouped parish level for confidentiality reasons. It is likely that parish boundaries will not coincide with catchment boundaries (Miles et al. 1996). Under such circumstances it is possible to assume that the proportion of agricultural activities in the parish (as a whole) is also present in the area of the parish within the catchment. Alternatively one can use the method of areal interpolation (Moxey and Allanson 1994).

There are a total of 6 parishes in the catchment (see map 6.1), most of which when viewed on a map lie outside the catchment boundary. Fortunately for this study farm level IACS (Integrated Administration and Control System) data for the West Peffer Catchment was made available by SEERAD provided it met confidentiality requirements. Access to farm level data is not usually allowed. Figures of the total area of land in the catchment under potato, wheat, spring barley, winter barley, oilseed rape, permanent and temporary grassland, setaside and protein pea cultivation were made available. In none of these 9 categories did the number of holdings fall below the minimum requirement of 5 holdings – a necessary condition for public disclosure.

Obviously farm boundaries do not necessarily coincide with catchment boundaries, but the error in using farm boundaries is likely to be considerably smaller than if parish boundaries are used. Unfortunately the data could not be refined further as the proportional method mentioned earlier (Miles et al. 1996) requires knowledge of farm areas which was not disclosed.

The sum acreage of all the crops reported (IACS statistics) in 1997 was approximately 4387 ha. Whereas the total catchment area by GIS estimation was approximately 4518 ha, and after subtracting buildings, loch and rough grazing (peaty and shallow soil in table 5.2) acreage left approximately 4346 ha of arable land – and was take to be the total catchment area in the model. The total acreage of each soil texture was estimated at sandy 876 ha, silty 732 ha and loamy 2740 ha.

Table 6.2: West Peffer Soil Classification

Peffer Soil Code	Area (metres square)	Area (ha)	Association	Series	Texture
4150400	9259605	925.9605	Kilmarnock	Kilmarnock	Loamy
7100500	3269857	326.9857	Rowanhill	Winton	Loamy
4151400	2362666	236.2666	Kilmarnock	Brownrigg	Loamy
1244000	400708	40.0708	Darleith	Darlskel	*shallow*
7101500	6634160	663.416	Rowanhill	Macmerry	Loamy
2302100	1204399	120.4399	Fraserburgh	Fraserburgh	Sandy
5002000	669074	66.9074	*Buildings*		
159400	1211774	121.1774	Alluvial soils	Ali	Loamy
4150432	1383288	138.3288	Kilmarnock	Kilmarnock	Loamy
7101532	408078	40.8078	Rowanhill	Macmerry	Loamy
1241100	70565	7.0565	Darleith	Darleith	Loamy
2302140	229617	22.9617	Fraserburgh	Fraserburgh	Sandy
1241144	343634	34.3634	Darleith	Darleith	Loamy
7100532	671294	67.1294	Rowanhill	Winton	Loamy
4153700	638471	63.8471	Kilmarnock	Loudoun	*peat*
1441400	6880163	688.0163	Dreghorn	Peffer	Silty
159900	1210207	121.0207	Alluvial soils	Alundiff	Loamy
1442100	7221495	722.1495	Dreghorn	Dreghorn	Sandy
1272100	96558	9.6558	Darvel	Darvel	Sandy
7249900	5046	0.5046	Saltings	Saltings	Silty
7490400	430121	43.0121	Stirling	Cauldside	Silty
5001800	12234	1.2234	*Loch*		
1241145	516805	51.6805	Darleith	Darleith	Loamy
159700	53000	5.3	Alluvial soils	Alp	Loamy
Total (ha)	**45182819**	**4518.282**			

The disclosed IACS figures revealed temporary grassland accounted for 43.04% of the total catchment grass production. Some winter barley is also cultivated, however as a proportion of the total catchment barley production it only accounts for 24.89%. However as the farm level statistics were made available after the nitrate leaching functions for specific crop rotations had been established spring barley and temporary grassland were not modelled.

If one ignores the difference between spring and winter barley and between temporary and permanent grassland then the catchment model's baseline (unrestricted crop allocation i.e. in the absence of regulation) crop allocation acreage is remarkably similar to farm level IACS figures (discussed in detail in the next chapter). However it must be noted that *differences between the model baseline and actual catchment land allocation are likely to be greater* due to a variety of reasons, some of which include a) protein peas were not modelled, b) temporary grass and winter barley was not modelled, c) some farm boundaries extend outside the catchment, c) IACS reporting errors, d) land use and soil type approximating errors, e) the assumption that farmer maximise utility by profit maximisation alone, f) capital cost constraints, g) the inability to model complex bio-physical considerations (such as pest control considerations, animal husbandry etc.). The following chapter discusses the difference between the model baseline and actual catchment land allocation in greater detail.

6.4 Crop Production functions

Based on experiments over two decades ADAS has compiled the results of a large number of field trials all over the UK (mostly England) relating artificial Nitrogen application with crop yield (Chambers and Johnson 1990). This data set was used to derive the crop production functions of five main crops.

A regression analysis of yield on nitrogen input using the a) linear, b) cubic and c) linear plus exponential (LpE) functional forms was carried out in MICROFIT (Pesaran and Pesaran 1991). The LpE form gave the best fit, see table 6.3. The choice of this functional form is consistent with previous work in the literature (George 1984; Lord and Mitchell 1998). The model uses the same crop production function irrespective of the previous crop. This is a modelling limitation as in reality yields are dependent on the previous crop (England 1986). The functions are convex in shape and have a maximum point within reasonable bounds of those found in the literature (England 1986)

The ADAS report (Chambers and Johnson 1990) uses the soil classification used by 'FERTIPLAN' (ADAS) and MAFF[70] which includes organic, peaty, shallow, other mineral, sandy, clay, deep silty, chalk and 'all other' soil categories. Obviously not every soil classification in the entire data set matched the sandy, silty and loamy categories; thus assumptions were made about classification of the available data on the basis of texture and drainage[71]. The derivation of potato growth functions in the presence of irrigation is discussed latter, as are the grass and silage production functions.

6.5 Nitrate Leaching

The nitrate leaching modelling comprised primarily of NITCAT (ADAS) and N-CYCLE which was developed by the Institute of Grassland and Environment Research (IGER). The nitrate leaching work was carried out by the Environmental Modelling and GIS group at ADAS (formerly the Agricultural Development and Advisory Service). The NITCAT model (Lord 1992; Lord et al. 1999) was used to estimate the relationship between nitrogen fertiliser application, arable crop management and nitrate leaching for all crop/soil combinations. NITCAT is an established model which has been used on behalf of MAFF and DETR to assess the impact of restrictions on agricultural practices within Nitrate Vulnerable Zones (NVZs) (Lord et al. 1999).

NITCAT requires detailed information on crop husbandry and rotations (sequence of crops grown, fertiliser use, manure inputs and timing, yield, harvest and sowing

[70] Reference booklet 209 'Fertiliser Recommendations for Agricultural and Horticultural Crops'
[71] This classification was performed with the help of Dr Allan Lilly (MLURI soil scientist).

dates, and ideally an estimate of the growth of autumn crops before winter sets in). Since it is generally agreed that farmer practices are varied, certain simplifying assumptions regarding crop husbandry were made, e.g. it was assumed that farmers applied two split nitrogen fertiliser applications (40kg/ha plus a variable).

Table 6.3 Crop Nitrogen Production Functions

Basic functional form: $y = \alpha_1 + \alpha_2(0.99)^N + \alpha_3 N$

Crop	Soil type	α_1	α_2	α_3	R^2	Maximum Physical Yield t/ha	Optimum Fertiliser Rate kg/ha - 1997/98 prices
Winter Oilseed Rape	Loam	3.501 (0.07344)	-1.523 (0.10919)	-0.0000795 (0.2307E-4)	0.91321	3.451	172.49
Winter Oilseed Rape	Silty	3.606 (0.00320)	-1.245 (0.00476)	-0.000190 (0.1008E-4)	0.99970	3.507	148.43
Spring Barley	Loam	7.242 (0.00634)	-4.013 (0.07353)	-0.00543 (0.3297E-03)	0.99880	5.615	137.80
Spring Barley	Sandy	7.554 (0.05959)	-4.292 (0.06908)	-0.0104 (0.3097E-03)	0.99807	5.035	104.66
Spring Barley	Silty	8.781 (0.07865)	-5.22 (0.09118)	-0.009615 (0.4088E-03)	0.9985	6.197	129.46
Winter Wheat	Loam	7.7716 (0.00594)	-2.972 (0.00738)	-0.000305 (0.2606E-4)	0.99997	7.601	181.47
Winter Wheat	Silty	7.8623 (0.040624)	-3.277 (0.050483)	-0.001278 (0.1782E-3)	0.99866	7.320	172.90
Winter Wheat	Sandy	8.295 (0.01269)	-4.341 (0.01614)	-0.00242 (0.5513E-4)	0.99992	7.357	182.98
Grass (grazing)	Loam	6.311 (0.15347)	-4.545 (0.26197)	0.00929 (0.3668E-3)	0.98839	3.911	Na
Grass (grazing)	Sandy	5.3194 (0.14313)	-4.284 (0.2443)	0.00817 (0.321E-3)	0.98769	3.1542	Na
Grass (grazing)	Silty	7.0225 (0.16917)	-4.6533 (0.28878)	0.01029 (0.4043E-3)	0.98780	4.442	Na
Grass (cutting)	Silty, Loamy, and Sandy	$y = 3.565 + 0.1279N$ (1.94192) (0.00810)			0.99601	Na	Na

Note: 1) there is no winter oilseed rape grown on sandy soils, nor are potatoes grown on silty or loamy soil, thus the corresponding entries are not present in table, 2) standard errors are in parenthesis, and 3) there is no one maximum nitrogen application rate for grass and silage as there is no unique output price for grass, These crops were combined in two 4-5 year rotations. The two rotations were a) *spring barley, winter wheat, spring barley followed by potatoes* on sandy soils and, b) *winter wheat, winter wheat, spring barley, followed by winter oilseed rape* on the remaining two soil categories. After consultation with ADAS it was decided that since the majority of grass production is on long-term grass (rather than ley-arable rotations) and as estimating nitrate loss when leys are ploughed out is difficult and inaccurate (Lord 2001b), it was assumed that grass is grown on permanent pastures only and thus not included in crop rotations. Permanent swards also have a number of potential benefits associated with sward density, which enable production to be utilized in conditions when leys would be vulnerable to poaching. There may be some nutritional benefits arising from species diversity in permanent swards, e.g. higher concentration of certain minerals (Hopkin 2000).

Nitrate losses depend on both the crop and its management besides the *subsequent crop and its management* especially during the autumn/early winter. For example if spring barley is followed by oilseed rape, nitrate leaching is smaller than if followed by winter wheat (which is drilled latter) or by bare ground kept free of weeds all winter until the next spring barley crop is sown.

Table 6.4: Range of Fertiliser Application tested

Crops	Min	Max	Survey of Fertiliser Practice (1998) Mean
Spring Barley	70	200	104
Winter Wheat	100	230	190
Winter Oilseed Rape	100	230	184
Man crop Potatoes	140	270	197
Grass Grazing	25	375	136*
Grass Cutting	25	375	136*

* Refer to long term or permanent grassland

NITCAT was not run for specific calendar years because the amount of excess winter rainfall (a key determinant of nitrate runoff) depends on both the previous crop and the whole winter of interest. NITCAT was run using metrological data which cuts across calendar years.

Metrological Office Data (including rainfall, evapotranspiration, max and minimum daily temperature) from 3 synoptic weather stations (Haddington, Stobshiel reservoir and Fordel Dean) for the period 1989-1998 in the neighbouring Tyne catchment was forwarded to ADAS, besides information regarding catchment soil textures, cropping and animal husbandry practices. Daily rainfall data for the West Peffer was assumed to be the average of the surrounding three metrological stations, see table 6.5.

Table 6.5: Annual Rainfall Data

Weather Type	Annual Rainfall (mm)	Reference Potential Evapotranspiration[72] (mm)
Mean	672	564
Wettest	827	530
Driest	470	602

The above data were used in regressions (Barrie et al. 1981) previously developed by ADAS which relate rainfall, mean PE, crop and soil type to the mean HER (hydrologically effective rainfall i.e. excess winter rainfall), see table 6.6 and 6.7. These regressions have been well substantiated across the UK and give the expected mean value for the results of MORECS (Meteorological Office Rainfall and Evaporation Calculation System) modelling (Thompson et al. 1981). The results for each soil type are presented below.

[72] Potential Evapotranspiration (PE) is the water that would evaporate from the surface and transpired by plants if the supply of water unlimited. It is calculated from the mean monthly temperature, with corrections for day length. From *PE* minus precipitation an approximate index can be calculated of the extent to which the water available for plants falls short of the amount they are capable of transpiring.

Table 6.6: Excess Winter Rainfall Calculation – Sandy Soil

Crop	Wet	Mean	Dry
Winter Wheat	412	263	71
Spring Barley	423	272	78
Winter Oilseed Rape	416	266	74
Maincrop Potatoes	426	279	91
Grass (Grazing)	354	208	22
Grass (Cutting)	354	208	22

Partitioning Excess winter rainfall

Rainfall was distributed evenly on a weekly basis (beginning 1 August) using the monthly rainfall means. These weekly values were multiplied by the ratio of the wettest annual total to the mean annual total to give the weekly values for a 1 in 10 wet year; and similarly by analogous calculation for a 1 in 10 dry year.

Table 6.7: Excess Winter Rainfall Calculation – Silty and Loamy Soil.

Crop	Wet	Mean	Dry
Winter Wheat	381	229	35
Spring Barley	396	224	47
Winter Oilseed Rape	396	243	47
Maincrop Potatoes	403	254	63
Grass (Grazing)	343	193	1
Grass (Cutting)	343	193	1

The NCYCLE model was used to estimate relationships between nitrogen fertiliser application, grass management and leaching (Scholefield et al. 1991). NCYCLE was run for both cutting and grazing grassland. The application of manure to grassland is discussed later.

The output from the IRRIGUIDE model (Anthony et al. 1996; Bailey and Spackman 1996a; Bailey and Spackman 1996b) was used to give crop-dependent weekly values of evapotranspiration over winter. It was assumed that the soil was at field capacity[73] by mid January. The following week in which evapotranspiration exceeded the rainfall was the week in which drainage ceased. Excess rainfall (rainfall – evapotranspiration) was then accumulated until the week in the autumn or winter at which the accumulated total was equal to the climatically expected excess rainfall for the crop. Thus an averaged value for weekly drainage and their start and end date for the mean, wet and dry scenario was obtained.

The temporal distribution of drainage, i.e. the actual week in which drainage begins and ends, varies substantially between years. However the relationship between cumulative drainage and nitrate concentration is relatively insensitive to the date of drainage, i.e. the first drainage events will generally carry the greatest concentration and consequently the concentrations will decline over winter according to the volume of water which as flowed rather than the date (Lord 2001b).

On freely draining soils such as sands and loams, such smooth behaviour is common. However on clays large flows tend to be of lower concentration because the rate cannot be accommodated by the clay matrix and most of the water tends to move through cracks to drains or overland –sometimes called 'bypass' or 'crack' flow. It does not have time to equilibrate with the soil matrix. Concentrations during low flow periods will tend to be greater than the mean while those during high flow periods will be smaller. In general, on such soils the mean concentration is about 60-80% of the peak concentration (Anthony et al. 1996; Lord 2001b). Fortunately the clay content of soils in the West Peffer is negligible.

Elution[74] behaviour was modelled using the SLIMMER algorithm (Anthony et al. 1996), which sums up the behaviour of more detailed models such as those of

[73] The percentage of water remaining in the soil 2 or 3 days after the soil has been saturated and free drainage has practically ceased. The percentage may be expressed in terms of weight or volume. The water content and the hydraulic conductivity of the porous medium (soil) is so low, that the water does not appear to 'flow or drain' downwards under the unit gradient of gravity, and is held by capillary forces and surface tension (Mayhew 1997).

[74] The process of removing an adsorbed material (adsorbate) from an adsorbent by washing it in a liquid (eluent) (Press 1999).

Addiscott and Burns. Essentially it relates how the quantity of nitrate leached is a diminishing function of the cumulative excess rainfall and of the soil water capacity, i.e. the greater the flow the greater the proportion lost. When the flow through the soil exceeds about 1.3 times the volume of water held in the soil in winter, the soil will have lost most of its nitrate (Scholefield et al. 1996).

By applying this equation to the potential nitrate loss calculated by NITCAT an estimate of the nitrate loss each week was obtained for each scenario i.e. each crop/soil/weather combination. NITCAT was run for each soil and climate (9 combinations) a total of 56 arable crops (4 rotation years multiplied by 14 N levels) plus 16 grass types (silage and cutting with 8 N levels).

The data set from these runs was regressed (OLS) to obtain estimated nitrogen leaching functions for crops and grass, table 6.8 and table 6.9 respectively. In order to determine the best 'fit' the adjusted R^2 and plotted residuals were compared in GENSTAT 5. The cubic (Lord and Mitchell 1998), semi-log (Sumelius 1994a), linear plus exponential and power functional forms were considered. The results showed that the power functional form gave the best fit by far.

Table 6.8: Crop Nitrogen Leaching Function

Functional Form: $\gamma_1 + \gamma_2 (N)^{\gamma_3}$

Crop	Previous Crop	Soil Type	Weather	γ_1	γ_2	γ_3	R^2
Winter Wheat	Spring Barley	Sandy	Dry	12.535 (0.115)	4.30E-15 (0.10E-13)	6.445 (0.429)	0.9998
Winter Wheat	Spring Barley	Sandy	Mean	39.423 (0.369)	1.20E-14 (0.29E-13)	6.462 (0.436)	0.9986
Winter Wheat	Spring Barley	Sandy	Wet	44.828 (0.408)	1.40E-14 (0.32E-13)	6.462 (0.426)	0.9977
Winter Wheat	Winter wheat	Silty	Dry	5.742 (0.135)	7.80E-09 (0.16E-07)	3.684 (0.365)	0.9958
Winter Wheat	Winter Wheat	Silty	Mean	36.832 (0.890)	0.000000094 (0.183E-06)	3.571 (0.355)	0.9921
Winter Wheat	Winter Wheat	Silty	Wet	54.62 (1.35)	0.000000142 (0.284E-06)	3.566 (0.362)	0.9861
Winter Wheat	Winter wheat	Loam	Dry	6.0394 (0.0983)	6.00E-12 (0.14E-10)	5.004 (0.441)	0.9967
Winter Wheat	Winter Wheat	Loam	Mean	35.75 (0.555)	2.90E-11 (0.677E-10)	5.042 (0.427)	0.9942
Winter Wheat	Winter Wheat	Loam	Wet	49.327 (0.766)	4.20E-11 (0.99E-10)	5.029 (0.426)	0.9983
Spring Barley	Winter wheat	Sandy	Dry	12.769 (0.580)	0.0000112 (0.0000148)	2.639 (0.246)	0.9897
Spring Barley	Winter Wheat	Sandy	Mean	36.63 (1.70)	0.0000333 (0.0000448)	2.632 (0.250)	0.981
Spring Barley	Winter Wheat	Sandy	Wet	41.14 (1.92)	0.0000384 (0.517E-04)	2.678 (0.251)	0.9869
Spring Barley	Winter wheat	Silty	Dry	4.191 (0.199)	0.001885 (0.000799)	1.5792 (0.0763)	0.9852
Spring Barley	Winter Wheat	Silty	Mean	21.54 (1.02)	0.00893 (0.00387)	1.5914 (0.0781)	0.9814
Spring Barley	Winter Wheat	Silty	Wet	31.3 (1.41)	0.01223 (0.00516)	1.6005 (0.0760)	0.9839
Spring Barley	Winter wheat	Loam	Dry	4.204 (0.299)	0.0000448 (0.474E-04)	2.296 (0.196)	0.9991
Spring Barley	Winter Wheat	Loam	Mean	20.01 (1.41)	0.000167 (0.183E-03))	2.339 (0.203)	0.9988
Spring Barley	Winter Wheat	Loam	Wet	26.28 (1.90)	0.000236 (0.262E-03)	2.327 (0.206)	0.9964

Crop	Previous Crop	Soil Type	Weather	γ_1	γ_2	γ_3	R^2
Oilseed Rape	Spring Barley	Silty	Dry	4.983 (0.511)	0.00157 (0.00115)	1.641 (0.127)	0.9852
Oilseed Rape	Spring Barley	Silty	Mean	25.58 (2.40)	0.00739 (0.00506)	1.655 (0.119)	0.9758
Oilseed Rape	Spring Barley	Silty	Wet	36.95 (3.37)	0.01049 (0.00699)	1.658 (0.116)	0.9772
Oilseed Rape	Spring Barley	Loam	Dry	5.867 (0.520)	0.0000148 (0.202E-04)	2.503 (0.245)	0.9967
Oilseed Rape	Spring Barley	Loam	Mean	27.98 (2.42)	0.0000533 (0.749E-04)	2.55 (0.253)	0.9988
Oilseed Rape	Spring Barley	Loam	Wet	36.76 (3.18)	0.000076 (0.105E-03)	2.536 (0.249)	0.9996
Spring Barley	Potatoes	Sandy	Mean	31.48 (1.71)	0.0000438 (0.0000569)	2.632 (0.241)	0.9967
Spring Barley	Potatoes	Sandy	Dry	10.984 (0.607)	0.0000147 (0.196E-04)	2.639 (0.247)	0.9994
Spring Barley	Potatoes	Sandy	Wet	48.65 (0.57)	0.0000265 (0.587E-05)	2.96 (0.101)	0.9999
Winter Wheat	Oilseed Rape	Silty	Dry	5.4811 (0.0769)	0.50E-08 (0.21E-07)	3.508 (0.736)	0.9999
Winter Wheat	Oilseed Rape	Silty	Mean	35.610 (0.416)	0.24E-08 (0.10E-07)	3.984 (0.776)	0.9986
Winter Wheat	Oilseed Rape	Silty	Wet	52.896 (0.314)	0.16E-08 (0.74E-08)	4.128 (0.816)	0.9985
Winter Wheat	Oilseed Rape	Loamy	Dry	5.7671 (0.0652)	0.10E-13 (0.232E-13)	6.177 (0.407)	0.9999
Winter Wheat	Oilseed Rape	Loamy	Mean	34.103 (0.415)	0.69E-13 (0.17E-12)	6.152 (0.435)	0.9997
Winter Wheat	Oilseed Rape	Loamy	Wet	47.103 (0.565)	0.92E-13 (0.22E-12)	0.6159 (0.430)	0.9991

Table 6.9: Grazing Grass Nitrogen Loss Function

Functional Form: $\beta_1 + \beta_2(N)^{\beta_3}$

Soil Texture	Weather	β_1	β_2	β_3	R^2
Grazing Grass					
Sandy	Dry	0.7766 (0.0461)	0.00107 (0.000103)	1.5979 (0.0160)	0.998
Sandy	Mean	1.737 (0.376)	0.004092 (0.00494)	1.7023 (0.0202)	0.992
Sandy	Wet	2.274 (0.512)	0.005515 (0.000678)	1.7011 (0.0205)	0.987
Silty	Dry	0	0	0	na
Silty	Mean	2.113 (0.308)	0.00377 (0.000513)	1.656 (0.0227)	0.989
Silty	Wet	1.908 (0.432)	0.004569 (0.000583)	1.6973 (0.0213)	0.981
Loam	Dry	0	0	0	na
Loam	Mean	2.221 (0.338)	0.004194 (0.000513)	1.6741 (0.0204)	0.991
Loam	Wet	2.103 (0.447	0.005238 (0.000641)	1.6992 (0.0205)	0.987
Cutting Grass					
Sandy	Dry	0.3783 (0.0472)	0.000471 (0.000104)	1.6002 (0.0369)	0.999
Sandy	Mean	0.829 (0.168)	0.002286 (0.000268)	1.665 (0.0196)	0.982
Sandy	Wet	1.071 (0.243)	0.003104 (0.000391)	1.6624 (0.0211)	0.976
Silty	Dry	0	0	0	na
Silty	Mean	1.003 (0.143)	0.002007 (0.000275)	1.6267 (0.0229)	0.979
Silty	Wet	0.921 (0.186)	0.002488 (0.000296)	1.6643 (0.0199)	0.979
Loam	Dry	0	0	0	na
Loam	Mean	1.04 (0.144)	0.002264 (0.000257)	1.6424 (0.0189)	0.983
Loam	Wet	1.032 (0.256)	0.002871 (0.000407)	1.6651 (0.0237)	0.974

(There is negligible leaching from cutting and grazing grass on loamy and silty soils under the dry weather scenario)

6.6 Irrigation Modelling

The West Peffer catchment is extensively used for surface water extraction and is presently subject to abstraction controls (Fox 1999). The need for controls arises from the damaging effects of uncontrolled surface water extraction on river ecology, wildlife populations, and recreational use[75] (anglers etc.) The current practice involves stopping abstractions through licence suspension when river flow falls below the MAF one-day flow at specific gauging points (Crabtree et al. 2000).

This research relies heavily on a study undertaken by MLURI on the economic impact of irrigation controls on potato farming in the West Peffer and Tyne catchment (Crabtree et al. 2000). The study was commissioned by SEERAD and is evidence of the Scottish Executive's interest in '*good ecological status*' as outlined under the water framework directive (WFD).

Field surveys suggest there is considerable reservoir storage (lined and unlined) in the West Peffer. Therefore historic measurement of stream flows reflect the managed system i.e. with abstraction and storage. The study modelled naturalised flows using the DIY model (Dunn 1998) which uses a GIS approach to spatially categorise the catchment into 50x50m cells on the basis of physical properties such as rainfall, elevation, hill-slope, distance to stream and soil type. The model allowed stream flows to be calculated for any location in the catchment, by estimating the weighted contribution of each category. The natural flow rate was estimated from Jan 1989 – Dec 1998 on a daily time step basis.

IACS data of the location and areas of potato crops was made available for the MLURI report (not this research), the report assumed potato irrigation occurs at the nearest point on the digitally derived stream network thus enabling a spatial distribution of irrigation abstraction to be estimated. The discrepancy between measured (very close to no flow) and modelled naturalised flows (minimum of 0.03 m^3/s) was so great that it could not be attributed to modelling errors; thus concluding that indeed in the West Peffer there are considerable abstractions despite the present irrigation control order. For detailed hydrograms, modelling assumptions and methodology refer to the report. Possible errors in the river flow modelling include a) limitations in the groundwater modelling, b) heterogeneity in the soil hydrological process, and c) feedback effects of soil moisture on actual evapotranspiration.

By assuming the regulator was able to impose abstraction bans instantaneously depending on the weather, the report modelled MAF restrictions at the 98th (NE), 95th (NF) and 90th (NT) percentile flow level. The 95th one-day flow defines a flow that would be exceeded naturally 95% of the days in a year. These flow restrictions pertain to 'natural' flows, i.e. in the absence of abstractions. Thus a 90th percentile flow restriction imposes a tighter surface water abstraction control than the 95th percentile restriction; and likewise the 95th is more stringent than the 98th percentile flow restriction.

Overall the river flow modelling enabled calculation of the total volume of water available for potato irrigation over the summer period for a *particular weather condition* and *percentile river flow restriction* for a particular day. This was done by subtracting the naturalised river flow from the percentile flow at the 98th, 95th and 90th % level.

6.7 Potato Production and Leaching equations

The potato growth and quality model was developed at Cambridge University Farm and consists of an irrigation scheduling model which predicts yield on the basis of potential water use and accounts for the effect of irrigation on the incidence and severity of common scab. Thus not only does it model potato yield but also quality. The 'Maris Piper' variety was modelled because it occupies 47% of the potato crop in East Lothian. Date of emergence was estimated from two temperature driven models in the literature (Mackerron 1985; Firman et al. 1992). Potato growth from emergence to tuberization was modelled according to previous work by the authors (Stalham et al. in press). For details on data source and model assumptions regarding

[75] Landscape amenity quality may also be reduced. For studies detailing the economic valuation of

date of planting, soil temperature estimation, scab risk, irrigation scheduling, cumulative water use and soil moisture deficits see the report.

The MLURI report attempts to classify each year in the 1989 – 98 period as 'dry', 'average' or 'wet' on the basis of a) rainfall, b) evapotranspiration and c) irrigation requirement *during the summer period (June – August).* It concludes, ' *it proved impossible to use a singe season for 'dry', 'average' or 'wet' scenario, but the best way of categorising would appear to be irrigation demand'* (Crabtree et al. 2000).

It must be noted that the derived leaching functions were based on stylised wet, mean and dry 'annual' rainfall which cut across calendar years (August – August). Thus in linking the leaching and production function for potatoes (for each weather condition) with the MLURI reports' associated irrigation regime it became necessary to *assume* that:

a) The 1989 June-August period with 'average' summer rainfall and high evapotranspiration, and irrigation requirement corresponded with *dry weather* conditions.
b) The 1991 June-August period with 'average' rainfall and evapotranspiration but 'low' irrigation corresponded with *mean weather* conditions.
c) While the 1998 June-August season with the 'wet' rainfall and low evapotranspiration and irrigation requirement corresponded with *wet weather* conditions.

It should be noted that Dr Mark Stalham (Cambridge University) potato growth modeller for the MLURI report liaised with Eunice Lord (ADAS) to calibrate the potato growth (table 6.10) and leaching functions (table 6.11) under different irrigation regimes and weather conditions. The potato leaching data was generated by NITCAT and regressed in MICROFIT similar to the crops mentioned earlier.

Scab was controlled through irrigation. It was assumed that infected tubers with a scab severity greater than 3% of surface area would be rejected for high quality

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river flow ecological benefits see (Willis and Garrod 1995).

packing; and if the proportion of scabbed tubers exceeded 30%, packers would not be able to differentiate and the sample would be rejected for a lower price market (processing or bag trade). The percentage of scab per hectare of produce for all combinations of weather, river flow restriction and irrigation regime is listed in table 6.10. Scab is the only differentiating aspect of tuber quality, other quality parameters such as cracking, secondary growth, bruising etc. are not considered.

Besides the option of not irrigating potatoes, the modelling process allowed for two irrigation regimes termed *optimum* and *restricted.* Of the two, optimum irrigation resulted in the better quality potato crop with significantly less incidence of scab. However, it must be noted that the *'optimum' strategy differs from full irrigation*, in that full irrigation prevents any loss of quality through scab infection, whereas optimum irrigation allows the maximum acreage of potato land to be irrigated with an acceptable level of scab. Optimum gives the best financial return, as the reduction in packable yield under optimal irrigation was outweighed by the larger tonnage of potatoes grown which met packing specifications.

Thus the potato crop may be divided into *scabbed* and *scab free*, the former being mostly associated with inadequate or 'restricted' irrigation, while the latter to a great extent with optimal irrigation. The difference is reflected in the market price of potatoes as a hectare of the 'optimally irrigated' crop which has a lower scab proportion and slightly greater yield is more profitable than a hectare of 'restricted' or 'un-irrigated' potato land. By restricting available irrigation water (through river flow controls) the regulatory authority lowers the profitability margin per hectare, prompting a shift in land allocation from optimal to restricted irrigation which reduces the incentive to apply as much nitrogen.

The potato and irrigation modelling yielded the total acreage of potato crops allowed under optimal and restricted irrigation for each river flow restriction in every representative year (see tables 6.12, 6.13, and 6.14). These upper bounds on acreage acted as constraints in the economic model reflecting the scarcity of irrigation water in each year. It must be noted that in the MLURI report, abstraction controls were not imposed to curb diffuse pollution but to simply ensure a minimum river flow.

Table 6.10: Potato Nitrogen Response Function

Functional Form: $\rho_1 + \rho_2(0.99)^N + \rho_3 N$

Weather Condition	Irrigation Regime	Flow Restriction	ρ_1	ρ_2	ρ_3	R^2	Scab (%)	Maximum Physical Yield (t/ha)
Dry	Optimum	NE	76.1 (0.0024)	-32.08 (0.0068)	-0.01993 (0.568E-4)	0.994	0	68.59
Dry	Optimum	NF	55 (0.0056)	-17.56 (0.0049)	-0.01993 (0.783E-3)	0.989	0	48.686
Dry	Optimum	NT	55 (0.0056)	-17.56 (0.0049)	-0.01993 (0.783E-3)	0.989	87.5	48.686
Dry	Unirrigated	NE	50.9 (0.00159)	-14.36 (0.0085)	-0.01993 (0.652E-4)	0.989	87.5	44.98
Dry	Unirrigated	NF	50.9 (0.00159)	-14.36 (0.0085)	-0.01993 (0.652E-4)	0.989	87.5	44.98
Dry	Unirrigated	NT	50.9 (0.00159)	-14.36 (0.0085)	-0.01993 (0.652E-4)	0.989	87.5	44.98
Mean	Optimum	NE	63.4 (0.00098)	-26.24 (0.00791)	-0.01993 (0.367E-3)	0.987	3.1	56.29
Mean	Optimum	NF	63.4 (0.00098)	-26.24 (0.00791)	-0.01993 (0.367E-3)	0.987	3.1	56.29
Mean	Optimum	NT	63.4 (0.00098)	-26.24 (0.00791)	-0.01993 (0.367E-3)	0.987	3.1	56.29
Mean	Restricted	NE	60.6 (0.0046)	-23.73 (0.00427)	-0.01993 (0.592E-3)	0.981	15.6	53.68
Mean	Restricted	NF	60.6 (0.0046)	-23.73 (0.00427)	-0.01993 (0.592E-3)	0.981	15.6	53.68
Mean	Restricted	NT	60.6 (0.0046)	-23.73 (0.00427)	-0.01993 (0.592E-3)	0.981	15.6	53.68
Mean	Unirrigated	NE	56.33 (0.0028)	-17.56 (0.00754)	-0.01993 (0.122E-3)	0.984	15.6	50.01
Mean	Unirrigated	NF	56.33 (0.0028)	-17.56 (0.00754)	-0.01993 (0.122E-3)	0.984	15.6	50.01
Mean	Unirrigated	NT	56.33 (0.0028)	-17.56 (0.00754)	-0.01993 (0.122E-3)	0.984	15.6	50.01
Wet	Optimum	NE	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	0	53.68
Wet	Optimum	NF	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	0	53.68
Wet	Optimum	NT	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	0	53.68
Wet	Restricted	NE	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	3.1	53.68
Wet	Restricted	NT	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	3.1	53.68
Wet	Unirrigated	NE	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	3.1	53.68
Wet	Unirrigated	NF	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	3.1	53.68
Wet	Unirrigated	NT	60.6 (0.0386)	-23.73 (0.00128)	-0.01993 (0.985E-4)	0.991	3.1	53.68

(No optimum N fertiliser rate is shown as the packable and scabbed potato yield fetch different market prices.)

Table 6.11: Potato leaching function

Functional Form: $\ell_1 + \ell_2(N)^{\ell_3}$

Weather Condition	Irrigation Regime	River Flow Restriction	ℓ_1	ℓ_2	ℓ_3	R^2
Dry	Optimum	NE	8.721 (0.914)	3.66E-05 (0.61E-13)	2.296 (0.637)	0.993
Dry	Optimum	NF	8.721 (0.914)	3.66E-05 (0.61E-13)	2.296 (0.637)	0.993
Dry	Optimum	NT	8.721 (0.914)	3.66E-05 (0.61E-13)	2.296 (0.637)	0.993
Dry	Unirrigated	NE	2.76 (0.879)	0.00244 (0.31E-12)	1.693 (0.797)	0.987
Dry	Unirrigated	NF	2.76 (0.879)	0.00244 (0.31E-12)	1.693 (0.797)	0.987
Dry	Unirrigated	NT	2.76 (0.879)	0.00244 (0.31E-12)	1.693 (0.797)	0.987
Mean	Optimum	NE	38.94 (0.987)	3.10E-09 (0.961E-11)	4.217 (0.936)	0.976
Mean	Optimum	NF	38.94 (0.987)	3.10E-09 (0.961E-11)	4.217 (0.936)	0.976
Mean	Optimum	NT	38.94 (0.987)	3.10E-09 (0.961E-11)	4.217 (0.936)	0.976
Mean	Restricted	NE	38.42 (0.571)	5.90E-08 (0.143E-11)	3.719 (0.912)	0.981
Mean	Restricted	NF	38.42 (0.571)	5.90E-08 (0.143E-11)	3.719 (0.912)	0.981
Mean	Restricted	NT	38.42 (0.571)	5.90E-08 (0.143E-11)	3.719 (0.912)	0.981
Mean	Unirrigated	NE	30.87 (0.217)	1.01E-05 (0.85E-10)	2.835 (0.865)	0.979
Mean	Unirrigated	NF	30.87 (0.217)	1.01E-05 (0.85E-10)	2.835 (0.865)	0.979
Mean	Unirrigated	NT	30.87 (0.217)	1.01E-05 (0.85E-10)	2.835 (0.865)	0.979
Wet	Optimum	NE	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975
Wet	Optimum	NF	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975
Wet	Optimum	NT	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975
Wet	Restricted	NE	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975
Wet	Restricted	NF	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975
Wet	Restricted	NT	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975
Wet	Unirrigated	NE	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975
Wet	Unirrigated	NF	46.26 (0.964)	7.10E-08 (0.759E-11	3.719 (0.951)	0.975
Wet	Unirrigated	NT	46.26 (0.964)	7.10E-08 (0.759E-11)	3.719 (0.951)	0.975

Table 6.12: Dry Weather Potato Growth and Irrigation Modelling

Details	Optimal 98	Optimal 95
Area (ha)	13	3
Cum water use (mm)	340	245
Stress (mm)	14	107
Irrigation (mm)	160	60
Drainage (mm)	2	6
Yield (t/ha)	81.6	58.8
Harvested yield (t/ha)	73.4	52.9
<40 mm yield (t/ha)	5.8	4.2
Saleable yield (t/ha)	67.6	48.7
Days >13.7 mm during scab control	0	0
Scab incidence (%)	0.0	0.0
Packable yield (t/ha)	67.6	48.7
Scabbed yield (t/ha)	0.0	0.0
Irrigation cost (£/ha)	640	240

Irrigation Scheduling		Optimal 98	Optimal 95
1st Irrigation	amount (mm)	15	15
	date	23-May	23-May
2nd Irrigation	amount (mm)	15	15
	date	30-May	01-Jun
3rd Irrigation	amount (mm)	15	15
	date	12-Jun	10-Jun
4th Irrigation	amount (mm)	15	15
	date	19-Jun	19-Jun
5th Irrigation	amount (mm)	25	25
	date	30-Jun	
6th Irrigation	amount (mm)	25	
	date	12-Jul	
7th Irrigation	amount (mm)	25	
	date	24-Jul	
8th Irrigation	amount (mm)	25	
	date	05-Aug	

Table 6.13: Mean Weather Potato Growth and Irrigation Modelling

Details	Optimal 98	Restricted 98	Optimal 95	Restricted 95	Optimal 90	Restricted 90
Area (ha)	164	65	150	60	137	55
Cum water use (mm)	283	270	283	270	283	270
Stress (mm)	3	3	3	3	3	3
Irrigation (mm)	65	50	65	50	65	50
Drainage (mm)	0	0	0	0	0	0
Yield (t/ha)	68.0	64.7	68.0	64.7	68.0	64.7
Harvested yield (t/ha)	61.2	58.3	61.2	58.3	61.2	58.3
<40 mm yield (t/ha)	4.9	4.6	4.9	4.6	4.9	4.6
Saleable yield (t/ha)	56.3	53.6	56.3	53.6	56.3	53.6
Days >13.7 mm during scab control	1	5	1	5	1	5
Scab incidence (%)	3.1	15.6	3.1	15.6	3.1	15.6
Packable yield (t/ha)	54.6	45.3	54.6	45.3	54.6	45.3
Scabbed yield (t/ha)	1.8	8.4	1.8	8.4	1.8	8.4
Irrigation cost (£/ha)	260	200	260	200	260	200

Irrigation Scheduling		Optimal 98	Restricted 98	Optimal 95	Restricted 95	Optimal 90	Restricted 90
1st Irrigation	amount (mm)	15	25	15	25	15	25
	date	24-May	27-Jul	24-May	27-Jul	24-May	27-Jul
2nd Irrigation	amount (mm)	25	25	25	25	25	25
	date	27-Jul	12-Aug	27-Jul	12-Aug	27-Jul	12-Aug
3rd Irrigation	amount (mm)	25		25		25	
	date	12-Aug		12-Aug		12-Aug	

Table 6.14: Wet Weather Potato Growth and Irrigation Modelling

Irrigation	Optimal 98	Restricted 98	Optimal 95	Restricted 95	Optimal 90	Restricted 90
Area (ha)	17	450	16	423	14	396
Cum water use (mm)	270	270	270	270	270	270
Stress (mm)	0	0	0	0	0	0
Irrigation (mm)	40	25	40	25	40	25
Drainage (mm)	66	52	66	52	66	52
Yield (t/ha)	64.8	64.8	64.8	64.8	64.8	64.8
Harvested yield (t/ha)	58.3	58.3	58.3	58.3	58.3	58.3
<40 mm yield (t/ha)	4.6	4.6	4.6	4.6	4.6	4.6
Saleable yield (t/ha)	53.7	53.7	53.7	53.7	53.7	53.7
Days >13.7 mm during scab control	0	1	0	1	0	1
Scab incidence (%)	0.0	3.1	0.0	3.1	0.0	3.1
Packable yield (t/ha)	53.7	52.0	53.7	52.0	53.7	52.0
Scabbed yield (t/ha)	0.0	1.7	0.0	1.7	0.0	1.7
Irrigation cost (£/ha)	160	100	160	100	160	100

Irrigation Scheduling		Optimal 98	Restricted 98	Optimal 95	Restricted 95	Optimal 90	Restricted 90
1st Irrigation	Irrigation amount (mm)	15	25	15	25	15	25
	Irrigation date	27-May	07-Jul	27-May	07-Jul	27-May	07-Jul
2nd Irrigation	Irrigation amount (mm)	25		25		25	
	Irrigation date	07-Jul		07-Jul		07-Jul	

6.8 Potato and Irrigation Modelling Assumptions and limitations

The MLURI report assumed a hypothetical soil, i.e. a clay loam/sandy clay loam/sandy loam soil with a 7% stone content. For the purposes of this research it was assumed that its properties were similar to those of a sandy soil. It should be noted that the potato growth and irrigation scheduling relied on rainfall measurements taken from the Haddington measuring station, while the rest of the crops were assumed to receive an average of all three surrounding metrological office stations Haddington, Stobshiel reservoir and Fordel Dean.

Another assumption is that the prices of outputs and inputs do not vary between years and stay constant at the 1997/98 level. The type of irrigation technology assumed in the study was a 100mm x 350 m raingun/hosereel; this was the most common form of irrigation technology in 1999-2000 (Crabtree et al. 2000).

The practicality of enforcing minimum river flow restrictions was not considered, for example the transaction costs (administrative and enforcement) of enforcing such regulation is not accounted for. Another practical consideration concerns the prediction of surface irrigation water demand, especially given the presence of reservoirs, boreholes, as it is likely the regulator will not have perfect information on irrigation technology, reservoir capacity and crop management etc.

The fact that catchments are not spatially uniform and don't have the same flow at all points i.e. low upstream flow. This implies that separate minimum acceptable flows (MAFs) and corresponding gauging points may have to be devised for different lengths of the river. In reality, a system of continuous monitoring and instantaneous response would probably not be plausible as farmers require notice and an erratic restriction might induce perverse behaviour by some farmers who over irrigate to compensate for an anticipated ban. A more appropriate control would entail an averaged restriction based on past weather patterns.

As only 8 of the 20 farmers in the West Peffer participated in the survey, the MLURI report had to *estimate* the reservoir and borehole capacity. These *estimates were not* *used in the potato modelling listed* in table 6.12, 6.13, and 6.14. because they were considered approximations.

6.9 Livestock Farming

Livestock farming is particularly difficult to model as herd structure and husbandry varies considerably and due to the presence of farm payments, such as beef special premium[76] and extensification premiums (based on stocking density), besides restrictions such as milk quotas. Transfer payment schemes and subsidy incentives for both livestock (SOAEFD 1997b) and arable cultivation (SOAEFD 1997a) were included in the model. In the discussion below where subsides are mentioned their eligibility conditions are assumed to be met. As a general caveat it should be noted that during the 1997/98 period beef markets were still greatly affected by the BSE crisis and the consequent ban imposed in Spring 1996. The prices budgeted in the study assume only a marginal recovery.

In total 5 types of livestock were modelled, diary cows, low ground suckler cows, over winter suckled calves, summer finished beef cattle and low ground sheep. Details of modelling assumptions pertaining to all 5 livestock types as well as production costs and market prices can be found in the *Farm Management Handbook 1997/98* (SAC 1997). Livestock numbers were converted into Grazing Livestock Unit (GLU) using standard conversion factors in the literature (Holmes 1989b; Nix 1997; SAC 1997).

Dairy cows were modelled as 'all year' cows producing an annual average of 6,000 litres of milk with a herd life of 4-5 years, winter feeding period 185 days, calving interval 390 days, and a cow size of 600kg approximately. The cow requirements or costs besides grazing grass and silage include concentrates[77], grainbeet, artificial insemination, vet and medicines, and other minor livestock expenses (SAC 1997). After considering these the gross margin before forage, i.e. before the cost of silage

[76] For example with beef special premiums male calves only are subject to a limit of 90 per head (SAC 1997)

and grazing production, is around £914. Concentrates are essential in maintaining a balanced diet and normally account for roughly 35%, 25% and 15% of the ME (metabolised energy) requirement of dairy cows, beef and sheep production (Forbes et al. 1980; M.A.F.F. 1984).

Low ground suckler cows were assumed to have a Feb-Apr calving period. After considering their relevant costs (i.e. barley and minerals, calf concentrate, purchased straw, vet and medicine, mart commission, haulage, bedding) and subsidies (suckler cow premium of £117.36 per cow) their gross margin before forage is £302 per head. This assumes the calf's weight at sale or transfer to winter finishing is 250kg. Similarly the gross margin before forage for *over wintered suckled calves* comes to £123 for steers and £73 for heifers (with finishing weight 385kg and 355kg respectively after 180 days feeding). While the same value for *winter finished beef cattle* is £77 and £71 per head for steers and calves respectively after 150 days of finishing. These are stylised statistics, for dairy-beef cross cows bred to a rand of mostly continental bulls, and a particular herd structure, mortality, replacement and housing system (SAC 1997).

It was assumed that sheep farming comprised of low ground breeding ewes with early finished lamb production (Suffolk crossed with Halfbred, Halfbred, Mule ewes with a terminal sire) averaging approximately140 finished lambs per 100 ewes. The lambs are sold at a live weight of around 40 kg and wool sales are included. The variable costs considered include barley and minerals, protein supplements, lamb concentrate, vet, drugs and dips, mart commission, haulage and shearing. The cost of purchased gimmers and replacement rams is also been included. The gross margin before grazing and silage production averages £4,781 per 100 ewes. This value includes the *sheep annual premium* at £13.3 per head. The catchment is not eligible to receive a *hill stock compensatory allowance*.

[77] In addition to grainbeet, hay and silage farmer provide protein, mineral and vitamin supplementation from conserved feeds (concentrates). Protein based supplements are used in forage-based diets, especially silages, for both dairy and beef cattle (Beever et al. 2000).

Given that only IACS figures on *suckler cows, bulls, beef cattle not for breeding, cattle under 1 year* and *total cattle* were available (other figures[78] could not be disclosed because they failed to meet the minimum requirement of 5 holdings) certain assumptions about herd structure were made to approximate the overall catchment stocking density at 2.2 GLU/ha. The additional *extensification premium* of £29.16 per head has been excluded. This is consistent with the observation that such premiums are normally awarded to *hill suckler cow* enterprises with stocking densities below 1.4 GLU/ha. The current estimated livestock numbers form the upper bounds in the model because they reflect the limited availability of inputs such as labour and capital costs and also the presence of farming restrictions such as milk quotas.

Stocking rate influences both individual animal performance and animal output per hectare. It is expressed as the number of animals per ha for a given time period. As stocking rate is increased, grazing pressure[79] increases, herbage allowance[80] decreases and the level of competition between animals increases. Consequently although herbage intake and animal performance decreases, the efficiency of herbage utilization (proportion of herbage removed relative to that available) increases (C.S. et al. 2000).

It was assumed that dairy and beef cattle are kept indoors during the winter period for 6 months, while sheep are kept indoors for 3 months (Holmes 1989b). During this wintering period the energy requirement of animals is predominantly met by silage and other supplements i.e. maize or barley silage, or barley and mineral supplements etc.

Initially a 'metabolised energy' (ME) approach was modelled (Forbes et al. 1980; M.A.F.F. 1984) which involved calculating the total ME requirement of each type of livestock (MJ/kg DM) and its equivalent dry matter content (g/kg). The dry matter

[78] The number of catchment dairy cows and heifers, beef heifers, other dairy cattle for breeding, other beef cattle for breeding and lamb statistics were not made available

[79] Grazing pressure is defined as the number of animals per unit mass of herbage.

[80] Herbage allowance is the weight of herbage DM (dry matter) per animal.

content is multiplied by the number of each livestock and equated with the dry matter content of grass and silage produced. However this procedure when compared with estimated statistics (SAC 1997) on the quantity of silage and grazing grass (yield) required to satisfy each GLU of livestock proved unnecessarily complicated and time consuming - it slowed the optimisation process and required the inclusion of additional constraints (bounds). The adopted procedure was also better because it allowed including other associated costs of livestock husbandry in a comparatively simpler manner. The results of both procedures were marginally different.

A linear silage production function, assuming 3 cuts, was estimated by regressing data from the Farm Management Handbook (SAC 1997) assuming a dry matter (DM) of 220 g/kg of settled silage. The model permits the option of purchasing silage. In reality there is marked *seasonal variation* in grass production, with maximum production in late spring and early summer. This seasonal surplus is harvested to make conserved feed for winter rations. Forage was traditionally conserved as hay, made in late summer, and is still an important feed, particularly for young stock. However since 1970's there has been a widespread adoption of silage as the major winter feed.

The aim of haymaking is to preserve grass or other forage by drying to a moisture content < 18%, at which point microbial and plant enzyme activity is minimal. Silage making (both bale and clamp) involves the additional use of chemical (acids) and biological additives (inoculants etc.) to conserve the protein content. Silage is generally described in terms of their pH values, the concentration of their fermentation end-products (lactic and volatile fatty acids) and their ammonia-N contents (Merry et al. 2000). Silage production also releases 'effluent'. Silage effluent has a biochemical oxygen demand of up to 90,000 mg oxygen per liter, which is almost 200 times more polluting than raw sewage, and even more polluting if released to water courses. It is regulated under the Control of Pollution Act 1992.

There is not much *rough grazing* land (peaty and shallow soil in table 5.2 amount to 103.91 ha) in the catchment and it is assumed to supplement sheep production. The approximate value of the number of ewes sustained by rough grazing is estimated by calculating the ME provided per ha of rough grazing - assuming an average rough grazing ME 6.5 KJ per kg DM of herbage production (Forbes et al. 1980; Holmes 1989a).

For both conserved feeding and grazing in situ, the feeding value of any forage depends on its ability to supply nutrients to the animal. This in turn has three main components: voluntary intake, nutrient content, and the animal's ability to absorb and utilize the nutrients. Whether directly or after storage (silage/hay) grass has to be utilized by animals before conversion into useful products; levels of utilized output are expressed in terms of energy, e.g. GJ/ha of utilized metabolizable energy (UME). 'UME output from grass shows large variations between farms, even for similar swards under similar inputs of fertilized N' (Forbes et al. 1980). Losses occur under grazing, particularly under poor drainage conditions, besides during harvesting, conservation and feeding out of the crop as hay or silage.

Grazing efficiency is defined as the proportion of herbage grown which is consumed by livestock and ranges from 50% to 90% (Holmes 1989a). Although typically a grazing efficiency of 70% is assumed (Holmes 1989b), after consulting with an East Lothian dairy and beef farmer an estimate of 78% was used.

Farm Yard Manure

Faeceas and urine from housed livestock are commonly managed as semi-liquid slurries or solid farm yard manures (FYM) and spread on to grassland as fertilizer. Livestock wastes are complex materials varying in physical, chemical and biological composition. They all contain water and wide range of carbon compounds, with appreciable amounts of plant nutrients and trace elements.

There are three potential sources of externalities from wastes. Firstly, a point-source of watercourses may result from, for example, burst or overflowing slurry stores. Secondly, diffuse pollution from leakages can occur, especially of plant nutrients, to both the aqueous environment (ammonia NH_3) and the atmosphere (greenhouse gasses e.g. nitrous oxide (N_2O) and methane (CH_4). Leaching N losses during storage of FYM depend on where and how it is stacked over winter. Thirdly gaseous losses or *volatilisation* to the atmosphere varies considerably depending on environmental factors and management. Thus estimating the N available from FYM is not considered an 'exact science' (SAC 1992). The availability of nitrogen in manure can vary from 50% to 75% while losses from volatilisation can range from 20% to 60% (Ryden 1984).

Besides the EU Nitrate Directive there are national or regional recommendations and regulations for controlling pollution in the EU. Most of these consist of licensing requirements for housed animals, minimum periods of storage for manure, and prohibited periods and methods of application to land. These are tailored to the requirements of individual countries.

Estimates of the N content of farm yard manure (FYM) and slurry from cattle and sheep were estimated (pigs and poultry farming in the catchment is negligible in comparison and excluded) assuming farmers comply with the *code of good practice* 'Prevention of Environmental Pollution from Agricultural Activity' (SOAFD), or the PEPFAA code, in all aspects of collection, storage and land application of FYM. The code stipulates procedures for land application, such as rate and uniformity of spreading, so that nutrient loss to water bodies is minimised.

Scottish Agricultural College (SAC) guidance or 'technical note' practices and figures were assumed as their recommendations are circulated widely among farmers. Estimates of the N '*available*' to the crop in the season of application for each type of livestock (*weight adjusted*) were calculated based on SAC technical notes (SAC 1992). The total N content of FYM comprises of *available* and *organically bound* N. The organic fraction is not available in the year of application and only of practical significance if large amounts are applied annually (some commentators assume that 20% of the remaining non-available organically bound N is released in each of the three years after application (Lord 1992)).

It should be clarified that although *casual* labour costs are included in the analysis, field and mobile machinery, fuel, building upkeep, drainage, fencing, miscellaneous costs etc. have not. Thus most labour charges and capital costs were not considered.

Setaside

The option of *voluntary setaside* land was included at a gross margin of £207.11 per ha including area payments. The catchment is not designated a less favourable area (LFA) hence non-LFA arable area payments apply. Obligatory setaside land was modelled with a gross margin of £301 per ha. It is assumed that the setaside land undergoes natural regeneration and thus seed costs (such as those of ryegrass or white clover) are not included.

An estimate of N leaching from setaside land was generated from the estimated grazing grass N loss function assuming natural regeneration of grass. In keeping with good agricultural practice it is assumed that setaside land is not temporary or ploughed as this releases lots of potentially leachable N.

Nitrate Modelling Summary

To summarise NITCAT essentially determined the amount of nitrate leaving the root zone and subsoil for all crop, soil, weather, minimum river flow restriction and potato irrigation regime combinations. It was *assumed that the water from the root zone mainly leaves the soil via deep drains and goes directly to the river within a day or within a few days*. Hence data in the following chapter is presented as a daily average for every week.

In reality there would be *some* 'vertical' nitrate leaching to groundwater. If water spends a long time in an aquifer before entering the river as 'base flow' there will be an averaging of concentrations (on a flow-proportional basis) over periods of months to centuries. Thus water originating from groundwater tends to have a fairly stable concentration. However modelling groundwater leaching is notoriously difficult and requires complicated dynamic modelling of catchment specific hydrological and geological processes which was beyond the scope of this study.

Many rivers are a combination of what was modelled (i.e. direct drainage to rivers) plus a groundwater component. Thus in winter when flows are high most of the water will be quick-pathway water from the land, showing fluctuating concentrations. While summer low flow concentrations would mainly reflect 'base flow' from groundwater (Lord 2002).

Thus overall the model estimates the total nitrogen in the root zone of all arable crops. It is assumed that this is transported to the river by rainwater or irrigation water via deep drains or surface runoff, where is diluted with a ground water 'base flow' contribution with no N content. It was assumed that the base flow can be estimated by averaging summer flow. This 'heroic' assumption was made because estimating the overall river concentration was required. Unfortunately detailed river and ground water hydrological modelling was beyond the scope of the study.

6.10 Economic Model

A non-linear model was written in General Algebraic Modelling System (GAMS) (Brooke et al. 1998) and solved using the CONOPT 2 solver (Stolbjerg-Drud 1993). CONOPT 2 solver is designed for models with non-linear constraints and a few degrees of freedom (Stolbjerg-Drud 1993).The large model comprised of 27865 single equations and 9049 single variables, where single refers to the individual rows and columns in the generated matrix. The results were confirmed by using the MINOS 5 solver which yielded similar results within reasonable bounds (see appendix 6.1 for a brief mathematical formulation of the model).

The model essentially minimises the cost of achieving the environmental standard (i.e. the WFD or Nitrate Directive (91/616) - which sets an upper bound of 11.3 mg per litre of nitrogen or 50mg of nitrates in drinking water) in the catchment. The regulator's objective is to minimise the difference between the unrestricted catchment profit (i.e. in the absence of regulation) and the catchment profits under environmental regulation. As all prices are exogenously determined, the minimisation of abatement costs is equivalent to *maximising restricted profit* under regulation (Cornes 1992). Put simply the problem of minimising the income forgone due to the environmental standard Θ can be written as:

$$Min\sum_i C(e_i)$$

$$subject\ to\ \sum_i e_i \leq \Theta$$

$$where\ \ \sum_i C(e_i) = \sum_i \left\{\Pi_i(p,w,x) - \left(\hat{\Pi}_i(p,w,\hat{x})\right)\right\}$$

The function $C(.)$ is the catchment abatement cost, i.e. the difference between unrestricted $\Pi_i(.)$ and restricted profits $\hat{\Pi}_i(.)$ for each unit of land *i*. Where *p* is a vector of output prices, *w* a vector of input prices, *x* the unrestricted input use and $\hat{x}$ the restricted input use. A more detailed explanation of the modelling framework is presented in Appendix 6.1. Such a framework allows for input substitution between land and fertiliser, i.e. both the intensive and extensive margin effects of policies can be examined. However it cannot *accurately predict* changes in production decisions regarding the input set and crop mix, as labour and capital (hence technology) are not considered. Input substitution is an important aspect of environmental policy design.

'Understanding input substitution is critical. Alternative policy strategies will provide different incentives to agricultural producers that may cause them to change their input sets' (Bouzaher and Shorgen 1995).

However it should be noted that the above conclusion is based on modelling of pesticides which have many more close substitutes than nitrogen. In most empirical studies of environmental regulation the capital and labour costs are ignored and assumed not to affect the relative ranking of alternative policies.

6.11 Transfer Payments and Transactions Costs

It is assumed the regulator's objective is to ensure compliance with the environmental standard at least cost to society. The cost of environmental regulation comprises abatement and transaction costs. Where 'abatement cost' is the *reduction in catchment profit from reduced catchment wide application of nitrogen fertiliser* under a policy and 'transaction costs' are the administrative costs borne by regulator to ensure compliance (Beavis and Walker 1983b). Transaction costs include the *designing*, *running* and *monitoring* costs of policies (Stavins 1995). The inclusion of transaction costs would enable realistic policy ranking (allowing the comparison between economic and administrative efficiency) however whether reliable proxies exist is questionable. A recent paper (Kampas and White 2002) has used the closest available estimates by approximating the weighted average of EU Agri-Environmental Schemes (Agricultural-Committee 1997). Although it is safe to make some generalisations such as, the transaction costs of emission based policies are higher than input based ones, this study *ignores transaction costs on the ground that accurate values are unavailable at present.*

Although the model accounts for *existing* subsidies and grants, any transfer payments made to induce compliance with the environmental standard are ignored as they distort the economic comparison. Thus tax revenues are not included in abatement costs as they are transfer payments from firm/farm to regulator and do not contribute to economic welfare (Prato 1998). It is argued that transfer payments do not affect the performance of the regional economy as a whole (Hoel 1998). This is the basis of the difference between *'farm profit'* (with transfer payments) and *'resource profit'* (excluding transfer payments) in some of the tables listed in the following chapter. In practice, assuming that the purpose of regulation is to control pollution and not generate income through taxation, the regulator can rebate the taxation amount. The double dividend hypothesis argues that such tax revenues may alleviate the distortion of pre-existing taxes (Pezzy and Park 1998). However investigating this was beyond the scope and purpose of this research and so not considered.

The following chapter presents the baseline solution and results of the model.

Appendix 6.1: A Concise Mathematical Representation of the Model

Regulatory objective

$$\textit{Minimise}\;\; \Pi^{\varpi\kappa} - \sum_c \sum_s \left(p_c Q_{cs} - w^n n_{cs} l_{cs}\right) + \sum_i \sum_j h_{ij}^{\varpi\kappa} \rho_j + \sum_b a_b p_b - w^n \left(\sum_i \eta_i^{\varpi\kappa} \lambda_i^{\varpi\kappa} - \sum_t \sum_s \mu_{ts} g l_{ts} \right) - C \qquad \text{(EQ 1)}$$

Subject to:

Secondary Expenses:

$$C = \sum_c \sum_j L_c \upsilon_{c\tau} - \sum_b \sum_m a_b f_{bm} - \sum_t \sum_u G_t z_{tu} - \sum_i \sum_x \lambda_i^{\varpi\kappa} q_{ix}^{\varpi\kappa} \qquad \text{(EQ 2)}$$

Crop production: $$y_{cs} = \gamma_{cs}^0 + \gamma_{cs}^1 \left(\gamma_{cs}^2\right)^{n_{cs}} + \gamma_{cs}^3 n_{cs} \qquad \text{(EQ 3)}$$

Total output: $$Q_{cs} = y_{cs} l_{cs} \qquad \text{(EQ 4)}$$

Grass production: $$g_{ts} = \beta_{ts}^0 + \beta_{ts}^1 + \left(\beta_{ts}^2\right)^{\mu_{ts}} + \left(\beta_{ts}^3\right)\mu_{ts} \qquad \text{(EQ 5)}$$

Grass requirement constraint: $$\sum_b a_b G_{tb} = \sum_s (g_{ts} g l_{ts}) \qquad \text{(EQ 6)}$$

Livestock N (kg/ha): $$\sigma = \frac{\sum_b a_b \Lambda_b}{\sum_t \sum_s g l_{ts}} \qquad \text{(EQ 7)}$$

Total grassland N (kg/ha): $$\mu_{ts} = \sigma + o_{ts} \qquad \text{(EQ 8)}$$

Total grassland Nitrogen use: $$\sum_t \sum_s \mu_{ts} g l_{ts} \qquad \text{(EQ 9)}$$

Livestock stocking density rate: $$d = \frac{\sum_b a_b}{\sum_t \sum_s g l_{ts}} \qquad \text{(EQ 10)}$$

Stocking density constraint: $$d \le \hat{d} \qquad \text{(EQ 11)}$$

Potato Production: $$\varphi_i^{\varpi\kappa} = \varepsilon_{0i}^{\varpi\kappa} + \varepsilon_{1i}^{\varpi\kappa} + \left(\varepsilon_{2i}^{\varpi\kappa}\right)^{\eta_i^{\varpi\kappa}} + \varepsilon_{3i}^{\varpi\kappa} \eta_i^{\varpi\kappa} \qquad \text{(EQ 12)}$$

Potato Quality/ Irrigation: $$h_{ij}^{\varpi\kappa} = H(\varphi_i^{\varpi\kappa}, \lambda_i^{\varpi\kappa}) \qquad \text{(EQ 13)}$$

Potato irrigation constraints: $$\lambda_i \le \hat{\lambda}_i^{\varpi\kappa} \qquad \text{(EQ 14)}$$

Bounds on land allocation: $G_t = \sum_s gl_{ts} \leq \hat{G}_t$, $L_c = \sum_s l_{cs} \leq \hat{L}_c$ (EQ 15)

Land use constraints: $\sum_t gl_{ts} + \sum_c l_{cs} + u\sum_i \lambda_i^{\varpi\kappa} \leq T_s$ (EQ 16)

Crop Rotational Constraints: $\sum_t l_{ts} r_{ts} + \sum_c l_{cs} r_{cs} + \psi \sum_i \lambda_i^{\varpi\kappa} r_i \leq 0$ (EQ 17)

Crop Nitrate load (kg/ha): $e_{cs}^{\varpi} = \delta_{0cs}^{\varpi} + \delta_{1cs}^{\varpi} (n_{cs})^{\delta_{2cs}^{\varpi}}$ (EQ 18)

Livestock Nitrate load (kg/ha): $v_{ts}^{\varpi} = \theta_{0ts}^{\varpi} + \theta_{1ts}^{\varpi} (\mu_{ts})^{\theta_{2ts}^{\varpi}}$ (EQ 19)

Potato Nitrate load (kg/ha): $x_i^{\varpi\kappa} = \xi_{0i}^{\varpi\kappa} + \xi_{1i}^{\varpi\kappa} (\eta_i)^{\xi_{2i}^{\varpi\kappa}}$ (EQ 20)

Overall River concentration (mg/litre):

$$ {}^{w}\wp^{\varpi\kappa} = \frac{\sum_c \sum_s {}^{w}e_{cs}\,{}^{w}\Gamma_{cs}^{\varpi} l_{cs} + \sum_t \sum_s {}^{w}v_{ts}\,{}^{w}\Omega_{ts}^{\varpi} gl_{ts} + \sum_i {}^{w}x_i^{\varpi\kappa}\,{}^{w}\Delta_i^{\varpi\kappa} \lambda_i^{\varpi\kappa}}{{}^{w}R + \sum_c \sum_s {}^{w}\aleph_{cs} l_{cs} + \sum_t \sum_s {}^{w}\lambda\!\!\!^{-}_{ts} gl_{ts} + \sum_i {}^{w}\hbar_i^{\varpi\kappa} \lambda_i^{\varpi\kappa}} \quad \text{(EQ 21)} $$

Environmental Quality Constraint: ${}^{w}\wp^{\varpi\kappa} \leq \Theta$ (EQ 22)

The regulator's objective is to minimise the difference between the unrestricted catchment profit $\Pi^{\varpi\kappa}$ and the catchment profit under different pollution control policies. Where ϖ is the prevailing weather condition that year (dry, mean, or wet) and κ is the catchment river flow restriction (no flow restriction or 98,95 and 90 percentile flow restriction) enforced by the regulator. $\Pi^{\varpi\kappa}$ for each $\varpi\kappa$ combination is the outcome of an unrestricted run of the model without any regulation. Thus when considering a particular regulatory policy it remains constant and independent of the optimisation problem. The catchment profit in the objective function is defined as the return to the producer's management and allocation of resources over the cost of total catchment nitrogen consumption $\{\sum_c \sum_s w^n n_{cs} l_{cs}$ (arable crops), $w^n \sum_i \eta_i \lambda_i$ (potatoes), $w^n \sum_t \sum_s \mu_{ts} gl_{ts}$ (silage and grazing grass)$\}$ and all other secondary costs of farming C. Where p_c is the market price of arable crop c, ρ_j the market price of potato quality j, and p_b is the market return from one grazing livestock unit (GLU) of livestock type b. w^n refers to the cost of nitrogen fertiliser, n_{cs} and l_{cs} is the nitrogen applied and land allocated to arable crop c (excluding potatoes and grassland) c on soil type s. gl_{ts} and μ_{ts} refer respectively to land and nitrogen allocated to grassland type t (grazing and cutting), while $\lambda_i^{\varpi\kappa}$ and $\eta_i^{\varpi\kappa}$ refer to land and nitrogen applied to potato crop under irrigation regime i (optimal, restricted or un-irrigated).

Secondary expenses C (EQ 2) refer to all other catchment production costs excluding that of nitrogen fertiliser application: $f_{bm} = (k_{b1},......k_{bm})$ is a vector of m costs per unit of livestock type (b) associated with feeding and other animal husbandry expenses, $z_{tu} = (\chi_{t1},....\chi_{tu})$ is a vector of u per hectare costs of grassland management, $\upsilon_{c\tau} = (v_{c1},......v_{c\tau})$ is a vector of τ per hectare costs associated with the production of each arable crop type and $q_{ix}^{\varpi} = \left(\omega_{i1}^{\varpi},......\omega_{ix}^{\varpi}\right)$ is a vector of x costs per hectare associated with potato farming including irrigation costs under each weather condition.

The crop production function equation set (EQ 3) yields the output (kg/ha) for each crop soil combination (the source of heterogeneity in the catchment) and is based on estimated coefficients $\gamma_{cs}^0, \gamma_{cs}^1, \gamma_{cs}^2, \gamma_{cs}^3$. The grassland yield for both silage and grazing grass on all soil types is given by the EQ 5, where $\beta_{ts}^0, \beta_{ts}^1, \beta_{ts}^2, \beta_{ts}^3$ are estimated coefficients. EQ 6 ensures that the actual grazing grass and silage production meets the requirements of livestock numbers a_b. If EQ 11 is satisfied then livestock qualifies for certain grants and subsides which are accounted for in p_b. EQ 15 is a constraint on the allocation of land, and ensures that the model allocation is similar to the actual situation on the ground. Most of these constraints were not binding. EQ 16 ensures the land allocation to any soil type does not exceed the actual acreage of each soil type. EQ 17 is a representation of the two representative rotational constraints in the catchment. As the model only allows potato allocation on sandy soils, $\psi = 0$ for silty and loamy soils and 1 for sandy.

EQ 12 is a set of equations for every weather (ϖ) and river flow restriction (κ) giving the potato yield per hectare under every irrigation regime i (optimal, restricted, and unirrigated) for nitrogen application η_i. Where $\varepsilon_{0i}^{\varpi\kappa}, \varepsilon_{1i}^{\varpi\kappa}, \varepsilon_{2i}^{\varpi\kappa}, \varepsilon_{3i}^{\varpi\kappa}$ are estimated coefficients for the potato production function. EQ 13, converts potato crop yield into quality categories j (scabbed and scab free), given the available irrigation water under each weather condition. EQ 14, limits the allocation of land to every irrigation category based on the available irrigation water.

EQ 18 estimates the total nitrogen load (per ha) E_{cs} for a total nitrogen application of n_{cs} (per ha) based on the weather estimated coefficients $\delta_{0cs}^{\varpi}, \delta_{1cs}^{\varpi}, \delta_{2cs}^{\varpi}$. Whereas EQ 19 and EQ 20 provide the annual load per ha from Livestock/grassland (V_{ts}) and potato (X_i) based on the weather estimated coefficients $\theta_{0ts}^{\varpi}, \theta_{1ts}^{\varpi}, \theta_{2ts}^{\varpi}$ and $\xi_{0i}^{\varpi\kappa}, \xi_{1i}^{\varpi\kappa}, \xi_{2i}^{\varpi\kappa}$ respectively. It is assumed that the nitrogen from animal waste allowed by MAFF regulation is applied to grassland. Λ_b is a vector of the estimated annual N content of one GLU of each livestock type. Therefore EQ 7 provides the per hectare availability of Nitrogen from animal waste to grassland, which along with the artificial N fertiliser o_{ts} provides the total Nitrogen application to grassland μ_{ts} per ha (EQ 8). The annual loads from EQ 18, 19, and 20 were converted into the average daily load for every week of a weather condition based on computations of NITCAT which gave three vectors. ${}^{w}\Gamma_{cs} = \left({}^{1}\alpha_{cs}^{\varpi}, \ldots\ldots, {}^{w}\alpha_{cs}^{\varpi}\right)$ is a proportionality vector of the average daily arable crop load for each week (w), ${}^{w}\Omega_{ts}^{\varpi} = \left({}^{1}\infty_{ts}^{\varpi}, \ldots\ldots, {}^{w}\infty_{ts}^{\varpi}\right)$ a proportionality vector of the average daily grassland/livestock crop load for each week , and ${}^{w}\Delta_{i}^{\varpi\kappa} = \left({}^{1}\vartheta_{i}^{\varpi\kappa}, \ldots\ldots, {}^{w}\vartheta_{i}^{\varpi\kappa}\right)$ a proportionality vector of the average daily potato crop load from each irrigation regime for each week.

Likewise the estimated daily average drainage (rainwater / rain + irrigation water) from each catchment activity (${}^{w}\aleph_{cs}^{\varpi}$ arable crops, ${}^{w}\lambda_{ts}^{\varpi}$ grassland, ${}^{w}\hbar_{i}^{\varpi\kappa}$ potatoes) for every week under all three weather conditions was calculated from the nitrate leaching model runs. EQ 21 gives the overall river concentration from farming activities at the mouth of the river assuming instantaneous mixing. ${}^{w}R$ is an approximation of daily river base flow for ever week under each weather condition. Unit conversions have been ignored in EQ 21. EQ 22 is the environmental constraint relating to river nitrate pollution, where Θ is the standard.

As the model was run for every weather condition and river flow control, the potato/irrigation variables, yield/leaching equations, and constraints varied accordingly. Similarly when a regulatory policy was considered corresponding adjustments to the constraints and constants were made.

Chapter 7
Results

7.1 Introduction

This chapter will present the results of empirical modelling outlined in the previous chapter. It is divided into an analysis of a) the model's baseline calibration, b) the presence of river restrictions (MAF) on instruments to control NPS nitrate pollution, c) changes in catchment land and fertiliser allocation under single and 'mixed' instrument NPS control policies, d) the comparative performance ranking of all policies under different weather conditions, and e) general conclusion.

7.2 Baseline Calibration

A comparison of the actual situation in the catchment as deduced from IACS data and the 'basic' baseline solution (i.e. without regulation) is presented in table 7.1. This table is termed the 'basic' baseline because it does not include the effect of river flow restrictions or different weather conditions. It is assumed that minimum acceptable flow (MAF) restrictions and weather affects the *break-up* of the total potato acreage into the 3 irrigation types, i.e. optimal, restricted and unirrigated – and therefore the catchment profitability.

In addition to the percentage deviation column in Table 7.1, two basic summary measures of how closely the bio-physical economic model replicates the baseline where calculated: a) Mean Absolute Deviation[81] (MAD), and b) Percentage Absolute Deviation[82] (PAD). For arable activities (grassland and setaside included) MAD = 83.58, while PAD = 20.58. Whereas for livestock MAD = 35.96 and PAD = 10.13. In calculating the statistics for livestock all animal numbers were converted to GLU (Grazing Livestock Units) for a meaningful comparison (SAC 1997). It must be noted that these statistics could only be calculated for activities for which the actual

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[81] Mean Absolute Deviation $MAD = \frac{1}{N}\sum_{i=1}^{N} | x_i - \overline{x} |$, where x_i are the N observations and $\overline{x}$ their mean.

[82] Percentage Absolute Deviation PAD is calculated in the same way as MAD except it uses percentage values instead.

catchment situation was known, i.e. disclosed values. Thus activities termed NA (Not Applicable) or 'not disclosed' in table 7.1 could not be included.

Table 7.1 Comparison between Actual Catchment Activities and Model Baseline Solution

	Actual Situation 1997/98 (IACS)	**Baseline Solution**	**Percentage Deviation**
Arable Area (ha)			
Potatoes	591.85	590	-0.31
Winter wheat	1968.61	1950.01	-0.94
Spring Barley	774.68	1030	32.96
Winter Barley	256.84	NA	NA
Winter Oilseed Rape	145.94	116.46	-20.20
Protein Peas	199.46	NA	NA
Total Arable	**3937.38**	**3686.47**	**- 6.37**
Grassland Area (ha)			
Permanent Grass	286.55	479.77	67.43
Temporary Grass	216.53	NA	NA
Total Grassland	**503.08**	**479.77**	**- 4.63**
Setaside (ha)			
Obligatory	**183.01**	**180**	**-1.64**
Livestock Numbers			
Suckler cows	384	400	4.17
Suckled calves	Not Disclosed	400	NA
Over-wintering calves	Not Disclosed	400	NA
Intensive Beef	Not Disclosed	180	NA
Bulls	15	NA	NA
Beef Cattle not for breeding	706	820	16.14
Cattle under 1 year	445	400	-10.11
Dairy cattle	Not Disclosed	300	NA
Ewes	Not Disclosed	50	NA
Total Cattle GLU	Not Disclosed	1106	NA
Stocking density	Not Disclosed	2.2	NA

7.3 Impact of Minimum Acceptable Flow (MAF) on NPS Control Policies

Section 7.3 will detail the modelled impact of minimum acceptable flow (MAF) restrictions on NPS control policies for the West Peffer catchment under different weather conditions, beginning with the *mean or average* catchment weather.

7.3.1 Mean Weather Conditions

The impact of a catchment artificial fertiliser tax (*nitrogen input tax*) was simulated by running the model iteratively with increasing nitrogen costs. Figure 7.1 shows the percentage increase in the price of nitrogen required to reduce the number of weeks in the year which exceed the standard with different river flow standards (MAF restrictions) under mean weather conditions. It is evident that with irrigation restrictions in place the required increase in nitrogen taxation is less than without any river flow controls and that the more stringent the surface water extraction control the lower the optimal tax. Secondly as the regulator tightens the requirement to meet the water quality standard (regulatory target) the greater the difference between taxation required with and without the river flow restrictions.

As detailed in previous chapters, the potato crop may be divided into *scabbed and scab free*, the former being mostly associated with inadequate or 'restricted' irrigation, while the latter, to a great extent, with 'optimal' irrigation. The difference is reflected in the market price of potatoes as a hectare of the 'optimally irrigated' crop which has a lower scab proportion and slightly greater yield is more profitable than a hectare of 'restricted' or unirrigated potato land. By restricting irrigation (through MAF controls) the regulatory authority lowers the profitability margin per hectare, prompting a shift in land allocation from optimal to restricted irrigation which reduces the incentive to apply as much nitrogen.

Regulatory Policies Under Different River Flow Conditions

Figure 7.1: Input Taxation

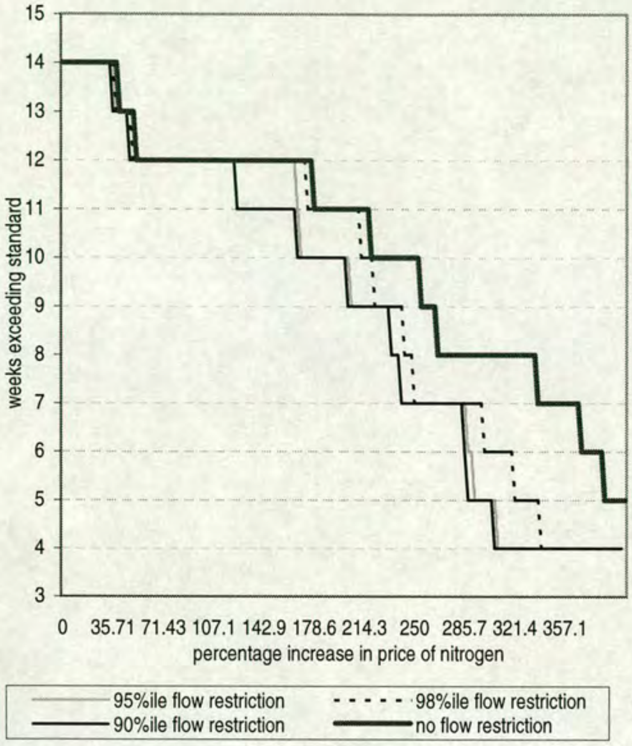


Figure 7.2: Stocking Density Reduction

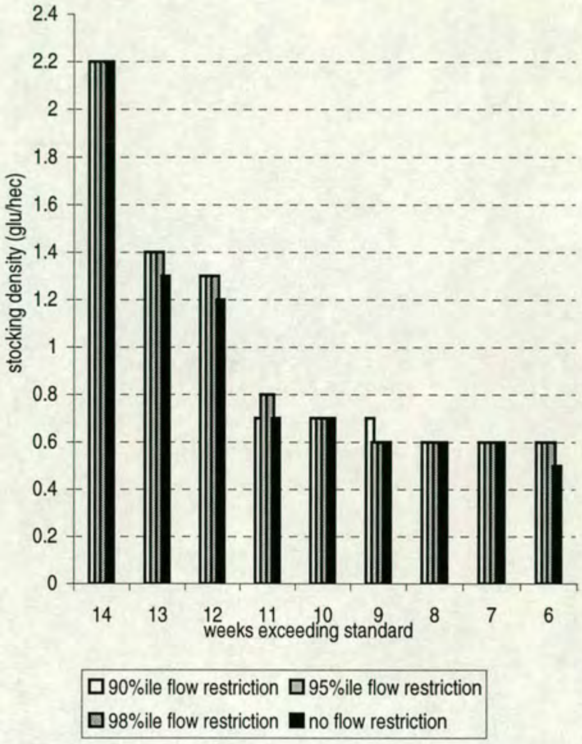


Figure 7.3: Setaside Reduction

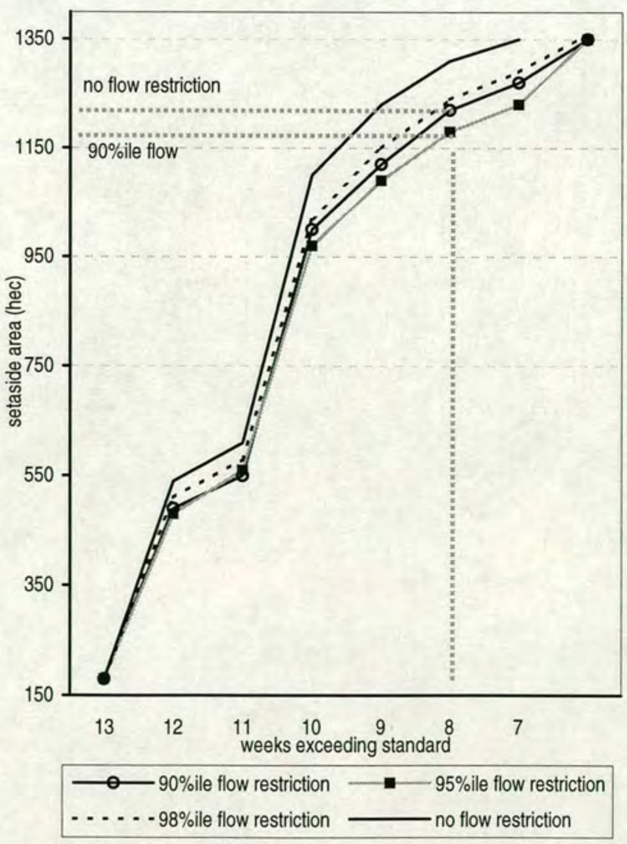
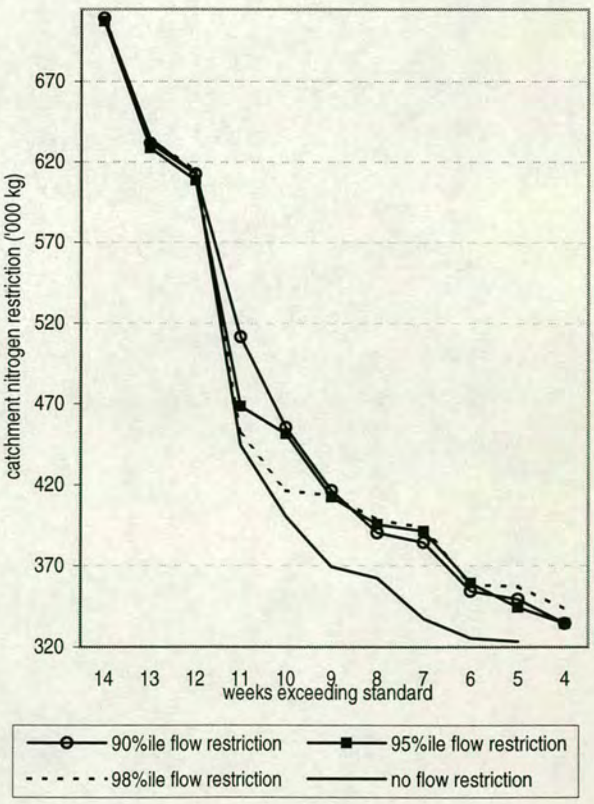
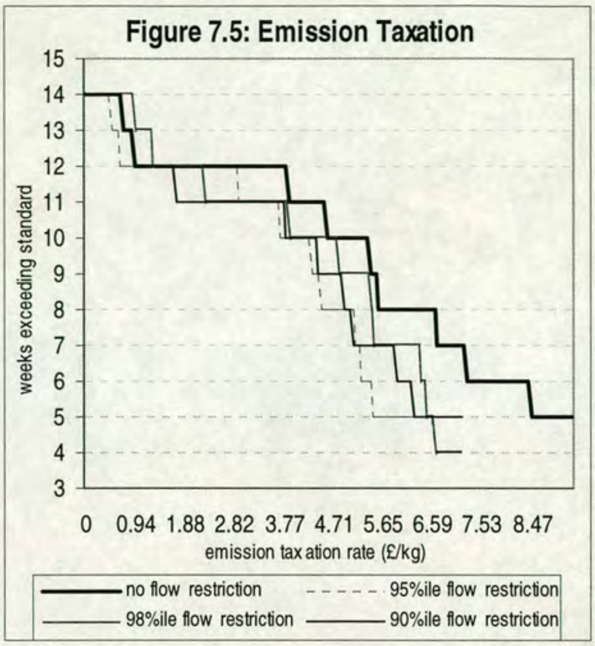


Figure 7.4: Nitrogen Input Quota





Four other measures to reduce diffuse nitrogen pollution under *mean weather conditions*. were considered. These were *stocking density reduction* (figure 7.2), *setaside*[83] *restriction* (figure 7.3), *input quota* (figure 7.4) and *emission taxation* (figure 7.5). The results are, for the most part, consistent with those for input taxation i.e. the presence of river MAF controls reduces the need to impose as stringent an instrument level to control diffuse nitrogen pollution when compared to the absence of any river flow restrictions. Under all regulatory regimes (NPS control policies) the distinction between the presence and absence of a river flow restriction is clear; the exception being stocking density reduction, where the difference is marginal.

However *the ranking amongst the three river flow restrictions is not consistent.* Irrespective of the pollution control policy, one would expect that the presence of the tightest river flow restriction (90th percentile) would required the least stringent diffuse pollution control policy followed by the 95th and then 98th percentile river flow restriction. The results are not entirely consistent in this regard due to certain rotational and livestock restrictions in the model. It seems that within the feasible region there are land, nitrogen and crop/soil allocations under the 95th and 98th percentile river flow restrictions which result in marginally less diffuse pollution than under the 90th percentile restriction. This is by no means an error; rather it highlights the complexity of bio-physical economic modelling.

[83] It is assumed that setaside land is not rotational. Rotational setaside is exacerbates diffuse nitrogen leaching.

7.3.2 Wet Weather Conditions

The same runs where carried out under the 'wet year weather' scenario where *the difference in diffuse nitrate pollution between any of the three minimum river flow controls and their absence was negligible* (figure 7.6). It is intuitively clear that irrespective of the river flow regime (MAF) and irrigation type (optimal, restricted or unirrigated) the leaching rates were fairly similar due to the high volume of water flow through the soil. Thus when rainfall is plentiful irrigation controls will not affect nitrogen input or land allocation as much. Therefore in terms of diffuse nitrate control in a wet year the presence of river flow restrictions has no real impact. The need to investigate various policies in the dry year did not arise as baseline runs showed pollution levels well below the standard.

Nevertheless, the question is whether it is more cost-effective to control diffuse pollution in the catchment with irrigation controls or by using conventional instruments which limit the polluting input's consumption. Suppose the regulatory target was to ensure that the standard was exceeded no more than 8 weeks of the year. In the absence of a river flow control (MAF) this required an input tax of 266.66%[84], whereas under a 95th percentile river flow control the required input tax was 233.33%. The extra resource cost (due to loss of productivity) from the higher tax under no river flow control amounts to £24,140 whereas the resource cost under the lower tax rate required with river flow controls is £901,954. *Therefore as a means to control diffuse pollution river flow controls alone are not an efficient*

[84] The market price of nitrogen in 1997/98 was £0.42 per kg.

mechanism. The reduction in pollution in the presence of river flow controls was not much when compared to the reduced crop profitability. Whether this can be said of other irrigated crops depends on the crop's input demand function for irrigation water (hence climate), production and nitrogen leaching function. It must be noted that this analysis does not consider the transaction/implementation costs of imposing and monitoring percentile bans on river flow which are likely to be higher than those of enforcing input taxation. Note that due to the inclusion of existing agricultural support payments profit overstate the net social benefits (similarly lost profits overstate recourse costs).

The impact of river flow controls on diffuse pollution depends on catchment specific MAF restrictions, regulatory targets, farming's dependence on surface irrigation water, weather, catchment hydrology and spatial aspects of extraction. In designing diffuse nitrogen pollution regulation in the presence of river flow restrictions the income distributional effects on irrigating and non-irrigating farmers will also need to be considered. Unfortunately the model assumed that potatoes were only grown on sandy soils. It would be interesting to compare the efficiency gains from river flow controls on potato leaching on other soil types.

7.4 Catchment Impact of NPS Control Policies

This section will detail catchment impact of different NPS control policies. In the tables which follow the term *catchment profit* refers to the 'farmer' profit in the presence of transfer payments such as taxes and setaside subsidies; whereas *resource profit* is the profit in the absence of any transfer payments. Thus 'farm profit' includes a setaside subsidy of £207.11 per ha (SOAEFD 1997a; SOAEFD 1997b), which is offered to farmers if they voluntarily setaside land in excess of their obligatory commitment under the Arable Areas Payment Scheme (AAPS), is assumed to be offered to catchment farmers under all NPS control policies considered. Similarly assuming the regional ceilings are not exceeded an extensification premium of £29.16 per head on both Beef Special Premium and Suckler Cow Premium is added *if* under any policy the stocking density falls below 1.4 GLU/ha and £42.12 if less than 1.0 GLU/ha (SAC 1997).

Two levels of standard compliance (or leniency of implementing the WFD 11.3mg/litre nitrate standard) are considered as policy targets i.e. one which permits the standard to be exceeded a maximum of 8 weeks of the year and one 6 weeks of the year. The catchment changes brought about by implementing a particular NPS control policy (i.e. instrument or instrument mixes) which ensures compliance with both regulatory target levels (i.e. 8 and 6 week standard target) are listed. The tables in the following sections will only consider NPS control policies under *mean* and *wet* weather conditions and with or without a NT (or 90th percentile) minimum acceptable river flow (MAF) restriction. Only the 90th percentile MAF restriction is considered as it represents the most stringent of the three river flow restrictions considered (see previous chapter). Since there are only very insignificant differences with and without MAF restrictions under wet weather conditions the two are not distinguished. Dry weather changes are not listed because nitrate losses to surface water under dry weather conditions are minimal and below the WFD nitrate standard.

The optimal NPS instrument(s) level under each weather/river flow scenario was determined by iterative increments in the instrument until compliance with the relevant policy target is achieved. *Setaside* in all tables refer to the sum of both *obligatory and voluntary setaside* land under the AAPS. The following subsections will detail the actual catchment changes in land allocation, fertiliser consumption, livestock numbers and profitability under each regulatory alternative. It must be noted that a) *all following table values are percentage changes relative to the baseline under each weather/MAF restriction*, and b) *that the baseline situation under each weather and MAF combination is different.*

7.4.1 Emission Based Policies

Two emission based policies (EBP) were examined - emission taxation and quotas. Emission taxation has often advocated in the economic literature because it is similar to pigouvian taxation. However, in reality, due to the very high monitoring costs associated with NPS pollution, emission based policies (EBP) are not viable. Instead regulators have relied on *estimated emission taxation* (Shortle and Dunn 1986;

Shortle and Abler 1994) which is based on biophysical economic simulation modelling, or an approach which combines the use of periodic actual emission measurements to calibrate and refine emission estimates generated from models. Biophysical simulation models express changes in resource quality as a function of management practices and location specific environmental practices (Weersink et al. 1998). ADAS combined both actual and estimates in monitoring and enforcing the NSA scheme (DEFRA 2002a). The potential legal and political difficulties in implementing an estimated emission taxation policy have been discussed in chapter 3.

Table 7.2 details percentage changes in catchment activities under estimated emission taxation. In terms of land allocation there is a shift away from arable farming and towards livestock production, thus the increase in grazing and silage land as well as an increase in livestock numbers. This is true of all weather/river flow combinations.

There is significant reduction in arable crop nitrate fertiliser consumption but not such a considerable reduction in grassland production, in fact in mean weather with the absence of river flow restrictions grassland application of fertiliser *increases* to support the increased livestock production. Overall there is significant catchment wide reduction in N application and a general shift to livestock production; this can be attributed to the fact that arable farming is relatively more polluting and less profitable than livestock farming. On comparing table 7.2 with table 7.3 (emission quotas) it is apparent that there is not much difference between the two. It is interesting to note that emission based policies are the only instruments which induce an increment in voluntary setaside land (under wet weather conditions). This is because setaside land is assumed to undergo natural grass regeneration without the application of fertiliser and thus produces relatively low nitrate emissions.

Table 7.2: Percentage Catchment Changes under Emission Taxation

Regulatory Target	Standard should Not Be Exceeded more than 8 weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	Mean Year		Wet Year	Mean Year		Wet Year
River Flow Restriction	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	-0.88	-2.35	-15.83	-0.88	-2.35	-56.23
Grazing land	5.38	15.52	54.93	6.31	15.52	47.33
Silage	6.47	14.21	57.39	4.04	14.21	39.67
Total Grassland (Silage + grazing)	5.69	15.16	55.61	5.69	15.16	45.21
Setaside	0	0	124.08	0	0	846.87
Fertiliser Use						
Potatoes	-15.0	-14.5	-21.4	-16.9	-16.8	-21.4
Arable Crops (excluding Potatoes)	-32.14	-34.21	-54.94	-34.48	-37.36	-81.92
Grazing Grass	-7.75	6.07	-27.53	-8.95	3.11	-29.86
Silage Grass	6.48	14.21	7.27	4.05	14.21	8.92
Total Grassland	-1.58	9.74	-12.68	-3.31	8.12	-13.31
Total Arable Crops (Arable Crops + Potatoes)	-28.92	-30.13	-48.20	-31.16	-33.09	-69.77
Total Catchment	-25.71	-25.71	-44.03	-27.89	-28.52	-63.14
Livestock						
Livestock Units (GLU)	6.33	15.37	15.37	6.33	15.37	15.37
Stocking Density	0	0	-21.82	0	0	-17.73
Profitability						
Farm Profit	-9.82	-10.02	-26.16	-11.62	-12.69	-26.82
Resource Profit	-0.95	-0.95	-3.92	-1.12	-1.21	-5.69

Table 7.3: Emission Load Quota

Regulatory Target	Standard should Not Be Exceeded more than 8 Weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	Mean Year		Wet Year	Mean Year		Wet Year
River Flow Restriction	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	-1.81	-2.35	-15.82	-2.34	-2.41	-56.43
Grazing land	11.83	15.52	54.97	15.52	15.52	47.32
Silage land	11.39	14.21	57.41	14.21	14.21	39.67
Total Grassland (Silage + Grazing)	11.71	15.16	55.62	15.16	15.16	45.23
Setaside (obligatory and voluntary)	0	0	124.81	0	0	845.87
Fertiliser Use						
Potatoes	-12.24	-14.51	-21.4	-14.29	-16.82	-21.41
Arable Crops (excluding Potatoes)	-24.53	-34.17	-54.94	-27.40	-37.40	-81.97
Grazing Grass	3.99	6.23	-27.57	4.82	3.10	-29.81
Silage Grass	11.39	14.21	7.27	14.21	14.21	8.72
Total Grassland	7.33	9.83	-12.68	9.05	8.12	-13.30
Total Arable Crops (Arable Crops + Potatoes)	-22.07	-30.09	-48.21	-24.78	-33.13	-69.75
Total Catchment	-18.47	-25.67	-44.03	-20.63	-28.56	-63.12
Livestock						
Livestock Units (GLU)	12.08	15.37	15.37	15.37	15.37	15.47
Stocking Density	0	0	-21.82	0	0	-17.73
Profitability						
Farm Profit	-0.95	-0.94	-3.92	-1.17	-1.22	-5.68
Resource Profit	-0.95	-0.95	-3.92	-1.17	-.21	-5.71

7.4.2 Input Based Policies

In practice the regulation of NPS nutrient pollution most commonly involves controls on artificial or natural nitrogen inputs (Anderson et al. 1990). Input taxation remains the most practical economic instrument to implement because of its relatively lower monitoring and enforcement costs since fertiliser sales are easier to record. Input taxation is often considered an imperfect substitute of pigouvian taxation.

Table 7.4 details the changes in the West Peffer catchment simulation model in the presence of nitrogen input taxation. Input taxation does not affect catchment land allocation as much as it does catchment nitrogen input application – generally the reduction in nitrate losses under input taxation occurs due to reduction in the intensity of farming and not its extensive margin. In fact, under an input tax level required to meet the policy target under mean weather conditions there is no change in any land allocation. There is an allocation of land away from arable farming towards livestock (i.e. grassland) production *only* under the much higher input taxation levels required to meet policy targets under wet weather conditions.

The most significant reduction in nitrate fertiliser application occurs in arable crops, whereas the reduction in grass production is insignificant in comparison. Arable crop fertiliser application rates reduce *over 50% and 85%* under mean and wet weather condition input tax levels respectively relative to the appropriate baselines. There is no change in livestock numbers or stocking density under mean weather conditions, this reduces under wet weather input tax levels. Interestingly even with over a 78% reduction in catchment nitrate fertiliser application, induced by wet weather input tax levels, the associated reduction in resource costs is less than 9% of the baseline. The difference between *farm* and *resource* profit reduction arise due to the tax amount which is excluded from the resource profit as it represents a transfer payment.

Table 7.4: Percentage Catchment changes under Input Taxation

Regulatory Target	Standard should Not Be Exceeded more than 8 Weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	**Mean Year**		**Wet Year**	**Mean Year**		**Wet Year**
River Flow Restriction	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	0	0	-4.83	0	0	-10.65
Grazing land	0	0	44.08	0	0	102.63
Silage	0	0	-2.68	0	0	-19.95
Total Grassland (Silage + grazing)	0	0	31.16	0	0	68.76
Setaside	0	0	0	0	0	0
Fertiliser Use						
Potatoes	-8.33	-7.93	-29.58	-10.01	-10.60	-53.95
Arable Crops (excluding Potatoes)	-51.47	-55.90	-99.47	-58.30	-67.93	-99.51
Grazing Grass	-1.29	-1.23	-42.03	-1.29	-1.23	-79.88
Silage Grass	-2.68	-2.68	-2.68	-2.68	-2.68	-19.95
Total Grassland	-1.89	-1.88	-24.92	-1.89	-1.88	-53.82
Total Arable Crops (Arable Crops + Potatoes)	-43.34	-45.95	-85.43	-49.21	-56.04	-90.35
Total Catchment	-38.49	-41.06	-78.44	-43.66	-50.03	-86.13
Livestock						
Livestock Units (GLU)	0	0	0	0	0	-10.85
Stocking Density	0	0	-23.64	0	0	-47.27
Profitability						
Farm Profit	-7.01	-7.20	-18.56	-8.20	-9.17	-29.76
Resource Profit	-1.53	-1.63	-8.65	-2.04	-2.62	-13.43

Table 7.5: Percentage Catchment changes under Input Quotas

Regulatory Target	Standard should Not Be Exceeded more than 8 Weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	Mean Year		Wet Year	Mean Year		Wet Year
River Flow Restriction	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	0	0	-4.23	0	0	-11.02
Grazing land	0	0	38.94	0	0	105.90
Silage	0	0	-3.17	0	0	-19.95
Total Grassland (Silage + grazing)	0	0	27.30	0	0	71.12
Setaside	0	0	0	0	0	0
Fertiliser Use						
Potatoes	-8.61	-8.97	-32.79	-10.21	-11.22	-52.36
Arable Crops (excluding Potatoes)	-52.72	-60.67	-99.47	-59.16	-70.12	-99.51
Grazing Grass	-1.32	-1.32	-30.38	-1.32	-1.32	-70.22
Silage Grass	-2.68	-2.68	-3.17	-2.68	-2.68	-19.95
Total Grassland	-1.90	-1.90	-18.55	-1.90	-1.90	-48.36
Total Arable Crops (Arable Crops + Potatoes)	-44.39	-49.89	-86.07	-49.92	-57.84	-90.03
Total Catchment	-39.32	-44.28	-78.27	-44.19	-51.30	-85.22
Livestock						
Livestock Units (GLU)	0	0	-0.31	0	0	-10.85
Stocking Density	0	0	-21.82	0	0	-48.18
Profitability						
Farm Profit	-1.60	-2.01	-9.34	-2.11	-2.89	-14.57
Resource Profit	-1.60	-2.01	-9.34	-2.11	-2.89	-14.57

Catchment changes under an 'input quota' NPS control policy are listed in table 7.5. The results are similar to input taxation; however input quotas are marginally more expensive. It should be noted that under quotas there is no difference between 'farm profit' and 'resource profit' reduction simply because no transfer payments are involved in the levying input quotas. The difference between input taxation and quota has been theoretically proved in the literature (Wu 1999; Wu and Babcock 2001).

7.4.3 Setaside Land

A voluntary setaside scheme was introduced under the MacSharry CAP reform primarily to reduce agricultural surpluses – however as it essentially removes land from cultivation it effectively reduces the total amount of nitrogen being applied in a catchment and so becomes an instrument of NPS nitrate pollution control (Burt and Haycock 1993). The impact of a setaside policy on the West Peffer catchment is presented in table 7.6. Under a setaside policy most of the cultivated land reduction occurs as arable land (excluding potatoes) primarily because such land is least profitable. Only under the much higher setaside levels required to achieve the policy targets under wet weather conditions are there reductions in grassland and increments in silage production, which nearly doubles (the only other increase in land allocation other than the setaside itself).

Fertiliser application on arable land is significantly reduced however total grassland applications are marginally affected, except under wet weather setaside levels where there is nearly a 18 % increase in total grassland fertiliser application. The grassland increase is required to keep the livestock numbers and stocking density constant, as land allocation to total grassland decreases.

Regarding profitability, since farmers are offered a subsidy to setaside further land *resource profits reduction is higher than farm profit reduction*. The most interesting result to be noted is the relatively low resource cost under setaside levels required to meet the regulatory targets under wet weather conditions.

Table 7.6: Percentage Catchment changes under Setaside

Regulatory Target	Standard should Not Be Exceeded more than 8 Weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	**Mean Year**		**Wet Year**	**Mean Year**		**Wet Year**
River Flow Restriction	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	-41.01	-43.60	-56.19	-44.89	-46.83	-59.42
Grazing land	0	0	-32.35	0	0	-25.15
Silage land	0	0	84.72	0	0	65.86
Total Grassland (Silage + grazing)	0	0	0	0	0	0
Setaside (obligatory and voluntary)	705.56	750.00	1066.67	772.22	805.56	1122.22
Fertiliser Use						
Potatoes	8.14E-06	-1.20E-05	-3.2E-06	6.34E-06	-5.60E-06	-1.6E-06
Arable Crops (excluding Potatoes)	-3.65E+01	-4.01E+01	-56.56	-4.07E+01	-4.35E+01	-59.89
Grazing Grass	0.00E+00	0.00E+00	36.45	-8.90E-11	0.00E+00	15.83
Silage Grass	1.26E-11	-1.00E-11	-7.87	0.00E+00	-1.00E-11	-6.12
Total Grassland	5.46E-12	-4.30E-12	17.54	-2.50E-11	-4.30E-12	6.46
Total Arable Crops (Arable Crops + Potatoes)	-29.62	-31.82	-45.231	-33.05	-34.51	-47.89
Total Catchment	-26.15	-28.12	-37.88	-29.18	-30.50	-41.53
Livestock						
Livestock Units (GLU)	0	0	0	0	0	0
Stocking Density	0	0	0	0	0	0
Profitability						
Farm Profit	-8.11	-7.68	-10.72	-8.98	-8.33	-11.34
Resource Profit	-11.33	-10.85	14.93	-12.50	-11.77	-15.77

7.4.4 Stocking Density Reduction

Stocking density reduction involves any measure which increases the amount of total grassland relative to the number of livestock GLU, i.e. an increase in total grassland or a decrease in livestock. The results of implementing a stocking density reduction are presented in table 7.7. As is expected the total grassland increases more than three times at the cost of a significant reduction in land under arable cultivation, however there is no change in potato cultivation acreage.

As a consequence there are significant cuts in the total grassland fertiliser application, both silage and grazing grass. Although there is a slight increase in livestock numbers overall there is a reduction in stocking density. Since the stocking density required to meet both regulatory targets under all weather/river flow restrictions is lower than 1.0 GLU/ha the catchment qualifies for an additional extensification premium which is the reason why farm profit are greater than resource profits. The model could not reach an optimal solution under the 6 week regulatory target by reducing stocking density reduction. This is primarily due to the numerous rotational constraints present in the model. It is safe to say that the stocking density required to satisfy the 6 week regulatory target had rotational constraints been removed would have been extremely low indeed – probably less than 0.4 GLU/ha.

Table 7.7: Percentage Catchment changes under Stocking Density Reduction

Regulatory Target	Standard should Not Be Exceeded more than 8 Weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	Mean Year		Wet Year	Mean Year		Wet Year
River Flow Restriction	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	No Feasible Solution
Arable (excluding Potatoes)	-34.86	-34.87	-48.04	-34.86	-48.04	
Grazing land	159.61	159.61	194.06	159.61	194.06	
Silage land	396.21	396.21	613.72	396.21	613.77	
Total Grassland (Silage + grazing)	224.98	224.98	310.03	224.98	310.03	
Setaside	0	0	0	0	0	
Fertiliser Use						
Potatoes	-4.50E-07	8.07E-08	1.48E-03	-4.50E-07	-4.40E-07	
Arable Crops (excluding Potatoes)	-2.96E+01	-2.95E+01	-4.49E+01	-2.96E+01	-4.43E+01	
Grazing Grass	-7.86E+01	-8.06E+01	-7.78E+01	-7.86E+01	-8.06E+01	
Silage Grass	-5.57E+01	-5.57E+01	-7.59E+01	-5.57E+01	-7.59E+01	
Total Grassland	-6.83E+01	-7.00E+01	-7.69E+01	-6.83E+01	-7.86E+01	
Total Arable Crops (Arable Crops + Potatoes)	-2.40E+01	-2.34E+01	-3.59E+01	-2.40E+01	-3.51E+01	
Total Catchment	-2.90E+01	-2.88E+01	-4.04E+01	-2.90E+01	-4.02E+01	
Livestock						
Livestock Units (GLU)	-1.09E+01	-1.09E+01	-1.08E+01	-1.09E+01	-1.08E+01	
Stocking Density	-7.27E+01	-7.27E+01	-7.73E+01	-7.27E+01	-7.73E+01	
Profitability						
Farm Profit	-1.15E+01	-1.04E+01	-1.42E+01	-1.15E+01	-1.42E+01	
Resource Profit	-1.16E+01	-1.42E+01	-1.45E+01	-1.16E+01	-1.94E+01	

7.4.5 Mixed Instrument Policies

Numerous mixed instrument NPS pollution control policies which combine economic and non-Economic instruments were considered. The relative performance of these mixed instruments relative to the individual instruments which comprise them is detailed below along with changes in catchment allocations and profitability – the relative ranking of all instruments is detailed in the next section.

7.4.5.1 Setaside and Input Taxation

Firstly a policy *combination of setaside land with input taxation* was imposed on the catchment. Ten different setaside levels were considered (in the range of 100 – 900 ha) along side an incremental input tax level until the regulatory targets were met under all weather/river flow conditions. Overall a mandatory setaside of 300 ha (in addition to the 180 ha obligatory setaside required under the AAPS) proved to be the most efficient level when combined with input taxation. The results of such a combination are presented in table 7.8.

Overall the increased land setaside is mostly at the expense of arable land (excluding potato land) and there is a nominal change in total grassland. However at input taxation levels required to meet compliance under wet weather conditions the reduction in arable cultivation is also accompanied by increased total grassland cultivation. There is considerable reduction in fertiliser applied to arable crops; while the overall reduction in total grassland fertiliser applications is considerably less (increasing under one condition). The number of livestock does not change and nor does the stocking density, the exception being at the higher input taxation required under wet weather conditions.

It should be noted that farm profit is greater than resource profit; since resource profit excludes compensatory setaside payments made to the catchment (those in addition to the obligatory setaside compensation required under the AAPS scheme) and also the additional cost of input taxation, both of which represent transfer payments. The comparatively low reduction in resource profits under input taxation levels required for wet weather conditions should be noted.

Table 7.8: Percentage Catchment changes under Setaside (300ha) and Input taxation

Regulatory Target	Standard should Not Be Exceeded more than 8 Weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	Mean Year		Wet Year	Mean Year		Wet Year
River Flow Restriction	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	-9.69	-9.69	-12.16	-9.69	-9.69	-18.70
Grazing land	0	1.02	23.05	0	1.028	81.43
Silage land	0	-2.68	-2.68	0	-2.68	-2.68
Total Grassland (Silage + grazing)	0	0	15.94	0	0	58.19
Setaside (obligatory and voluntary)	166.67	166.67	166.67	166.67	166.67	166.67
Fertiliser Use						
Potatoes	-4.95	-5.68	-23.55	-6.35	-7.73	-36.25
Arable Crops (excluding Potatoes)	-40.69	-48.69	-98.28	-47.22	-58.30	-99.55
Grazing Grass	-1.64	8.94	-30.17	-1.64	8.94	-61.10
Silage Grass	-3.2E-09	-2.68	-2.68	-3.2E-09	-2.68	-2.68
Total Grassland	-0.93	3.70	-18.44	-0.93	3.70	-36.17
Total Arable Crops (Arable Crops + Potatoes)	-33.96	-39.77	-83.31	-39.52	-47.82	-86.87
Total Catchment	-30.09	-34.95	-75.71	-35.00	-42.11	-80.93
Livestock						
Livestock Units (GLU)	0	0	0	0	0	0
Stocking Density	0	0	-13.64	0	0	-36.82
Profitability						
Farm Profit	-5.69	-6.58	-16.8	-6.75	-8.19	-22.27
Resource Profit	-2.91	-2.98	-9.28	-3.19	-3.53	-11.34

Table 7.9: Percentage Catchment changes under Setaside (700 ha) and Input tax

Regulatory Target	Standard should Not Be Exceeded more than 8 Weeks			Standard should Not Be Exceeded more than 6 Weeks		
Weather	**Mean Year**		**Wet Year**	**Mean Year**		**Wet Year**
River Flow Restriction	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	-22.61	-22.61	0.00	-22.61	-22.61	-3.14
Grazing land	0	0	1.03	0	0	22.70
Silage land	0	0	-2.68	0	0	-2.68
Total Grassland (Silage + grazing)	0	0	0	0	0	15.69
Setaside (obligatory and voluntary)	388.89	388.89	388.89	388.89	388.89	388.89
Fertiliser Use						
Potatoes	-2.72	-3.24	-14.87	-4.02	-4.56	-23.74
Arable Crops (excluding Potatoes)	-37.13	-37.69	-82.97	-43.80	-45.48	-98.62
Grazing Grass	3.39	0.00	-1.11	3.39	0.00	-24.86
Silage Grass	0.00	0.00	-2.68	0.00	0.00	-2.68
Total Grassland	1.91	0.00	-1.82	1.91	0.00	-14.85
Total Arable Crops (Arable Crops + Potatoes)	-30.65	-30.55	-66.84	-36.31	-36.99	-80.87
Total Catchment	-26.83	-27.16	-58.43	-31.83	-32.89	-72.34
Livestock						
Livestock Units (GLU)	0	0	0	0	0	0
Stocking Density	0	0	0	0	0	-13.64
Profitability						
Farm Profit	-6.28	-6.64	-10.81	-7.25	-7.71	-15.10
Resource Profit	-6.08	-7.90	-8.38	-6.24	-8.09	-11.20

When compared to a setaside restriction alone the mixed policy presents *a major improvement* in ensuring regulatory compliance cost-effectively. Although it does not present an improvement over input taxation under mean weather/river flow restrictions – *under wet weather the setaside/input taxation policy mix even outperforms input taxation.*

For comparison table 7.9 lists the changes in the catchment under a NPS control policy which combines a setaside of 700ha with input taxation. As a comparison of resource profit will illustrate, the 700ha setaside policy mix outperforms 300ha setaside policy mix under 'wet weather' conditions but not under 'mean weather' conditions.

7.4.5.2 Stocking Density Reduction and Input taxation

The second two instrument combination policy considered involved a mix of *stocking density reduction and input taxation.* Eight different stocking density rates were examined (2.1, 2.0, 1.8, 1.6, 1.4, 1.2, 1.0, 0.8) together with incremental input taxes iteratively; out of which a stocking density reduction of 1.4 GLU/ha proved to be most cost-effective - the results of which are presented in table 7.10.

Unlike the previous mixed policy involving setaside and input taxation there is a) a significant increase in grassland acreage at the expense of reduced arable crop land b) since there are no increases in livestock numbers there is a substantial reduction in stocking density – over 35%, and c) the application of nitrogen to grassland is significantly reduced. The difference between farm profits and resource profits is due to the extensification premium and input tax, both of which have been detailed earlier.

This mix performs *significantly better* than stocking density reduction alone, but not as good at input taxation alone, even though under some weather/river flow restrictions (particular at the 6 week regulatory target) it nearly equals input taxation in terms of cost-effectiveness.

Table 7.10: Percentage Catchment changes under Stocking Density Reduction (1.4 glu/ha) and Input Taxation

Regulatory Target	**Standard should Not Be Exceeded more than 8 Weeks**			**Standard should Not Be Exceeded more than 6 Weeks**		
Weather	**Mean Year**		**Wet Year**	**Mean Year**		**Wet Year**
River Flow Restriction	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90^{th} %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	-8.85	-8.85	-8.85	-8.85	-8.85	-6.15
Grazing land	78.96	78.96	79.98	78.96	78.96	61.89
Silage land	0	0	0	0.0075	0	-19.95
Total Grassland (Silage + grazing)	57.14	57.14	57.14	57.14	57.14	39.28
Setaside (obligatory and voluntary)	0	0	0	0	0	0
Fertiliser Use						
Potatoes	-3.15	-3.38	-22.70	-4.36	-4.65	-74.22
Arable Crops (excluding Potatoes)	-30.59	-35.37	-98.49	-37.26	-43.02	-99.48
Grazing Grass	-48.31	-53.18	-54.68	-48.31	-53.18	-64.06
Silage Grass	0.00	0.00	-2.68	0.00	0.00	-19.95
Total Grassland	-26.52	-30.49	-30.72	-26.52	-30.49	-43.73
Total Arable Crops (Arable Crops + Potatoes)	-25.43	-28.73	-83.29	-31.07	-35.05	-94.42
Total Catchment	-25.55	-28.93	-77.53	-30.55	-34.52	-88.86
Livestock						
Livestock Units (GLU)	0.00	0.00	0.00	0.00	0.00	-10.85
Stocking Density	-36.36	-36.36	-36.36	-36.36	-36.36	-36.36
Profitability						
Farm Profit	-4.78	-5.03	-15.63	-5.69	-6.07	-42.02
Resource Profit	-2.51	-2.29	-8.75	-2.69	-2.52	-17.96

7.4.5.3 Setaside and Stocking Density Reduction

A policy mix comprising of two non-economic instruments was also imposed on the catchment model. A setaside restriction of at least 700 ha combined with an incremental stocking density reduction until compliance with the two regulatory targets was achieved. Unfortunately such a policy mix offered only a minor improvement over a policy comprising of either stocking density reduction or setaside restrictions alone, therefore no table of catchment changes is presented.

7.4.5.4 Three Instrument Mix: Setaside, Stocking Density Reduction and Input tax

Numerous 3 instrument mixes were also considered, both comprising of an economic instrument (input taxation) and two non-economic NPS control policies (i.e. setaside and stocking density reduction). The model was run with 5 different setaside and 6 different stocking density reduction levels along side incremental input taxation to decide which combinations represented an improvement in cost-effective abatement.

A setaside land restriction of 300ha, 1.4 glu/ha stocking density reduction combined with input taxation was the most cost-effective combination (table 7.11). Such a policy resulted in at least a 57% increase in total grassland at the expense of arable cultivation (excluding potatoes which are not reduced). Since there is no increase in livestock numbers catchment stocking density is reduced. Nitrogen fertiliser application to all catchment activities is reduced, potatoes experience the least reduction, while the grazing grass the most under input tax levels required to meet compliance under mean weather conditions. Under wet weather taxation levels the most reduction is under arable cultivation.

When compared with a single instrument policy of either a setaside reduction or a stocking density reduction the 3 instrument mix presents a major improvement at both regulatory targets under each weather/river flow restriction. However this 3 instrument policy is *only better than input taxation alone under wet weather conditions*. The implications of this result are discussed in detailed in the following section.

Table 7.11: Percentage Catchment changes under a 3 Instrument Mix: Setaside (300 ha), Stocking Density Reduction (1.4 glu/ha) and Input taxation

Regulatory Target	**Standard should Not Be Exceeded more than 8 Weeks**			**Standard should Not Be Exceeded more than 6 Weeks**		
Weather	**Mean Year**		**Wet Year**	**Mean Year**		**Wet Year**
River Flow Restriction	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions	90th %ile River Flow Restriction	No River Flow Restriction	With or without River Flow Restrictions
Land Use						
Potatoes	0	0	0	0	0	0
Arable (excluding Potatoes)	-18.54	-18.54	-18.54	-18.54	-18.54	-18.54
Grazing land	78.96	78.96	79.98	78.96	78.96	79.98
Silage land	0.00	0.00	-2.68	0.00	0.00	-2.68
Total Grassland (Silage + grazing)	57.14	57.14	57.14	57.14	57.14	57.14
Setaside (obligatory and voluntary)	166.67	166.67	166.67	166.67	166.67	166.67
Fertiliser Use						
Potatoes	-1.65	-2.65	-13.88	-2.45	-3.54	-29.17
Arable Crops (excluding Potatoes)	-28.05	-30.99	-81.85	-33.05	-37.12	-99.55
Grazing Grass	-51.57	-48.28	-51.94	-51.57	-48.28	-50.59
Silage Grass	0.00	0.00	-2.68	0.00	0.00	-2.68
Total Grassland	-29.14	-26.50	-30.52	-29.14	-26.50	-29.75
Total Arable Crops (Arable Crops + Potatoes)	-23.07	-25.11	-68.20	-27.29	-30.16	-85.41
Total Catchment	-23.79	-25.27	-63.84	-27.50	-29.75	-78.98
Livestock						
Livestock Units (GLU)	0.00	0.00	0.00	0.00	0.00	0.00
Stocking Density	-36.36	-36.36	-36.36	-36.36	-36.36	-36.36
Profitability						
Farm Profit	-5.26	-6.00	-13.05	-5.87	-6.73	-19.60
Resource Profit	-4.85	-5.27	-7.38	-4.91	-5.37	-10.81

7.5 Relative Efficiency of Instruments

This section will compare the relative efficiency of all the instruments detailed above. Figures 7.7, 7.8 and 7.9 are diagrammatic representations of the percentage reduction in catchment profit under various policy measures to control pollution. To reiterate, percentage reductions from the *relevant weather/MAF baseline* were plotted as the actual baseline catchment *profit* varies in the presence of different MAF restrictions and weather conditions. Catchment profit was highest in a mean year with no river flow controls (£9,302,000), followed by the wet year (£9,095,000), and lastly the mean year with the 90%ile river flow restriction (£8,402,000). Lack of sunshine, not water, was the limiting factor in wet weather conditions.

The mixed policies are represented by dotted lines in figures 7.7, 7.8, and 7.9. Two levels of a) the setaside/input taxation policy mix (300ha and 700ha), and b) the 3 instrument policy mix (1.4 glu/ha, and 1.8 glu/ha) are considered and illustrated in the figures.

7.5.1 Relative Ranking Mean Year

Both mean year figures (7.7 and 7.8) illustrate the superiority of economic instruments, with and without river flow controls, when compared to managerial approaches such as livestock density reduction or setaside (excluding any wildlife/biodiversity benefits). Single instrument economic approaches also fare better when compared to mixed instrument policies. The exception is stocking density reduction/input taxation mix which out performs input taxation and quotas marginally at the 6 week regulatory target and onwards under instrument levels required in mean weather conditions without any flow restrictions. In fact a combination of the two non-economic managerial instruments performs better than either managerial approach on its own. The mean year figures show a combination of stocking density and setaside reduction is more cost-effective at meeting ambient pollution standards at both the 8 and 6 week compliance level than both stocking density and setaside restriction alone.

Instrument ranking in table 7.12 is similar to those of previous empirical studies. Uniform emission taxation is superior to other controls (Johnson et al. 1991)

Table 7.12: Policy Ranking Under Different River Flow Restrictions and Weather Conditions

Regulatory Target	Standard should not be Exceeded more than 8 Weeks			Standard should not be Exceeded more than 6 Weeks		
Weather	Mean Year Rank		Wet Year Rank	Mean Year Rank		Wet Year Rank
River Flow Restriction	90^{th} %ile river flow restriction	No river flow restriction	90^{th} %ile river flow restriction	90^{th} %ile river flow restriction	No river flow restriction	90^{th} %ile river flow restriction
Input Quota	3	3	6	3	4	5
Input tax	2	2	4	2	3	4
Emission Based Policies (Taxation and Quota)	1	1	1	1	1	1
Setaside Restrictions	8	9	9	9	8	6
Stocking Density Reduction	9	8	7	8	9	9
Optimal* Setaside, Stocking density reduction with input taxation	6	6	2	6	5	2
Optimal* Setaside and input taxation	5	5	3	5	6	3
Setaside and stocking density reduction	7	7	8	7	7	7
Stocking density* (1.4 glu/ha) + input tax	4	4	5	4	2	8

* refers to the best level of setaside and stocking density for that weather and river flow scenario.

and outperforms input taxation provided the emission function exhibits increasing returns to scale (Stevens 1988). An input taxation does better than a quota at all target levels stricter than 12 weeks (cross-over), a result which is likely if firm heterogeneity is present (Wu 1999; Wu and Babcock 2001); firm heterogeneity is represented by 3 soil types and 8 crops (2 grass and 3 potato types). Table 7.12, does not differentiate between emission taxation and quotas and jointly lists them as 'emission based policies'. The table also lists some mixed instrument policies as 'optimal', this refers to the fact that although different instrument levels (e.g. setaside or stocking density reduction) were considered it ranks those which performed best.

Targeted input or emission quotas were not considered. Stocking density reduction does better than setaside at lower levels of standard compliance but undergoes a 'cross-over' at higher levels. The cost-effective difference between economic and pure regulatory policies increases as the policy requirements to meet the standard are tightened. 'Land retirement' or setaside is known to impose higher abatement costs since emissions are primarily a result of land management and not land use (Kampas and White 2002).

The relative ranking of instruments in the mean year is by and large not affected by the imposition of river flow controls (compare figures 7.7 and 7.8). This is illustrated by considering a regulatory target which ensures the standard is not exceeded at least 8 weeks of the year represented by the vertical line in all three figures. In comparing single instrument polices the ranking is fairly consistent irrespective of the standard compliance level with the exception of the two 'cross-overs' noted above and one between two mixed instrument policies.

7.5.2 Relative Ranking Wet Year

Although *at instrument levels required to induce regulatory compliance under wet year conditions* emission taxation remains the most cost-effective policy, the relative efficiency of policies changes considerably (figure 7.9). In a wet year nitrogen leaching rates (emissions) are considerably higher than in the mean year. The wet year leaching baseline is approximately 18 weeks of river nitrogen levels in excess of the standard, compared to 14 in the mean year. So although in a wet year considered there is more drainage (subsoil percolation), the nitrogen content of leached emissions in some weeks is enough to offset the dilution.

A notable change is that economic controls targeting inputs i.e. input taxation and quota *do not* perform as well in wet conditions especially at high standard compliance levels (refer to the 6 week regulatory target - figure 9). In comparison setaside/setaside mix policies perform considerably better in wet weather conditions. The improved cost-effective abatement under setaside/setaside mix policies increases the tighter the level of standard compliance (consider the difference in catchment profit between input taxation and setaside/setaside mix polices at 10, 8 and 6 week standard compliance).

The change in policy ranking in wet conditions can be explained by the difference in the economic incentives offered by each instrument. While setaside removes land from agriculture input quotas and taxation do not. Although input taxation and quotas provide an incentive to decrease nitrogen use per hectare, nitrogen is still applied. Therefore the potential to leach in a wet year remains.

Regulation may set an input tax based on a mean year's leaching pattern believing the taxation rate is sufficient to reduce NPS nitrogen pollution to the required standard compliance level (regulatory target) *on average*, but when the weather is wetter than expected, more nitrate is leached and compliance is not achieved. To illustrate, the optimal tax level to ensure standard compliance at the 8 week regulatory target in mean weather conditions is a tax increasing input cost to 1.4 pounds per kg, this instrument under wet weather conditions would result in the standard being exceeded 13 weeks of the year. See Table 7.13 for a comparison of the difference in instrument levels required to achieve compliance under different river flow restrictions and weather conditions.

Figure 7.7: Mean Year With No Flow Restriction

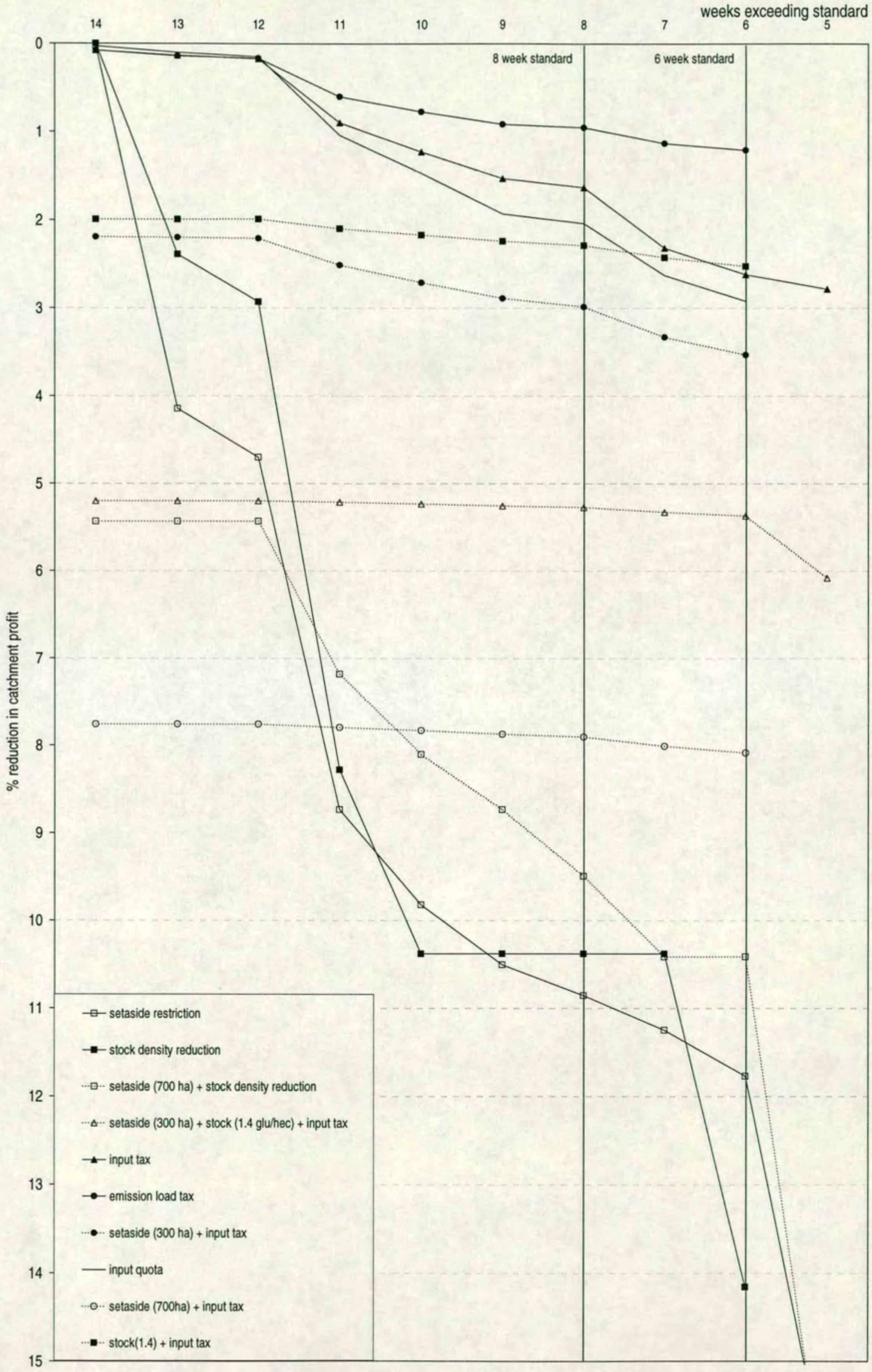


Figure 7.8: Mean Year With 90%ile Flow

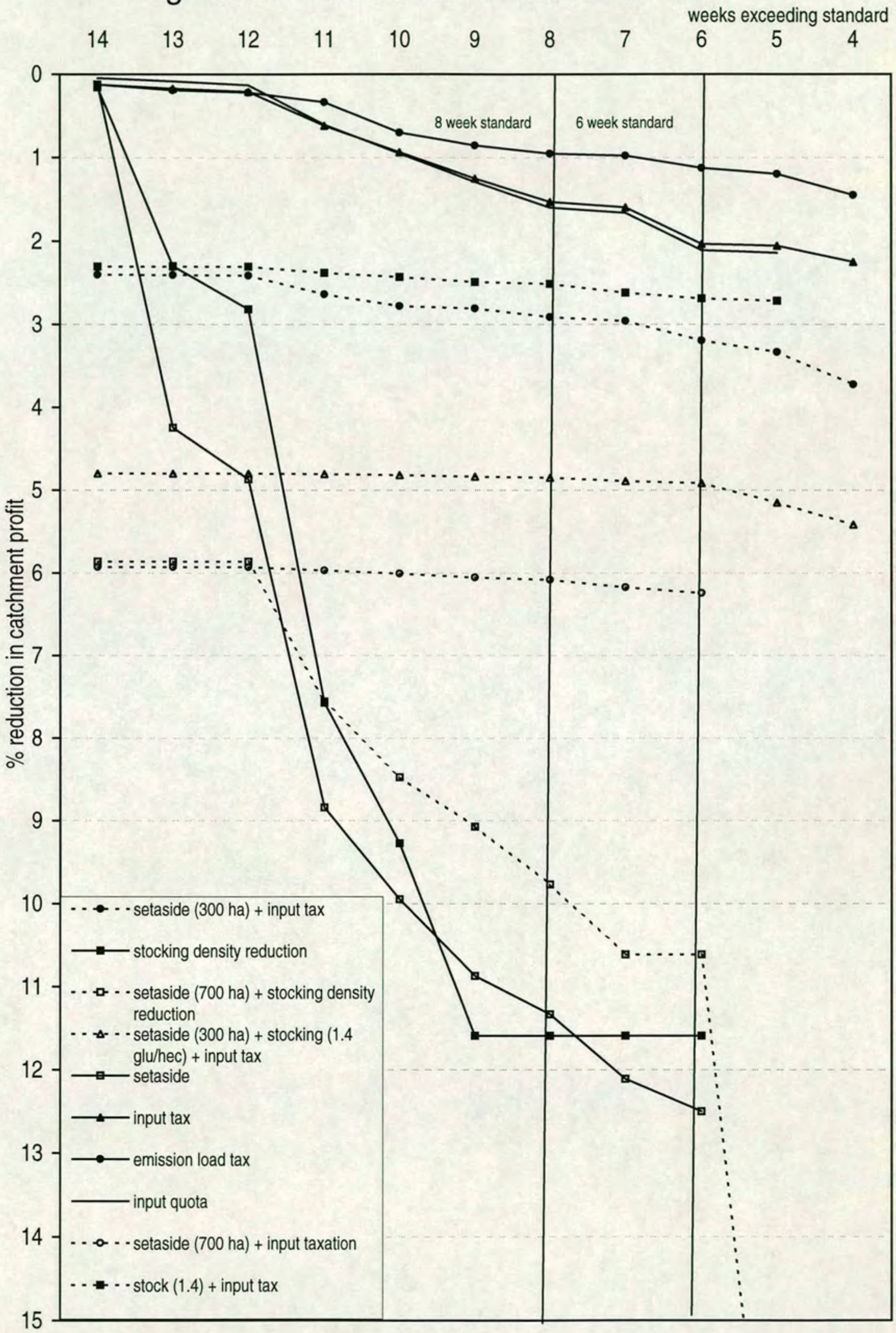


Figure 7.9: Wet Year With or Without Flow

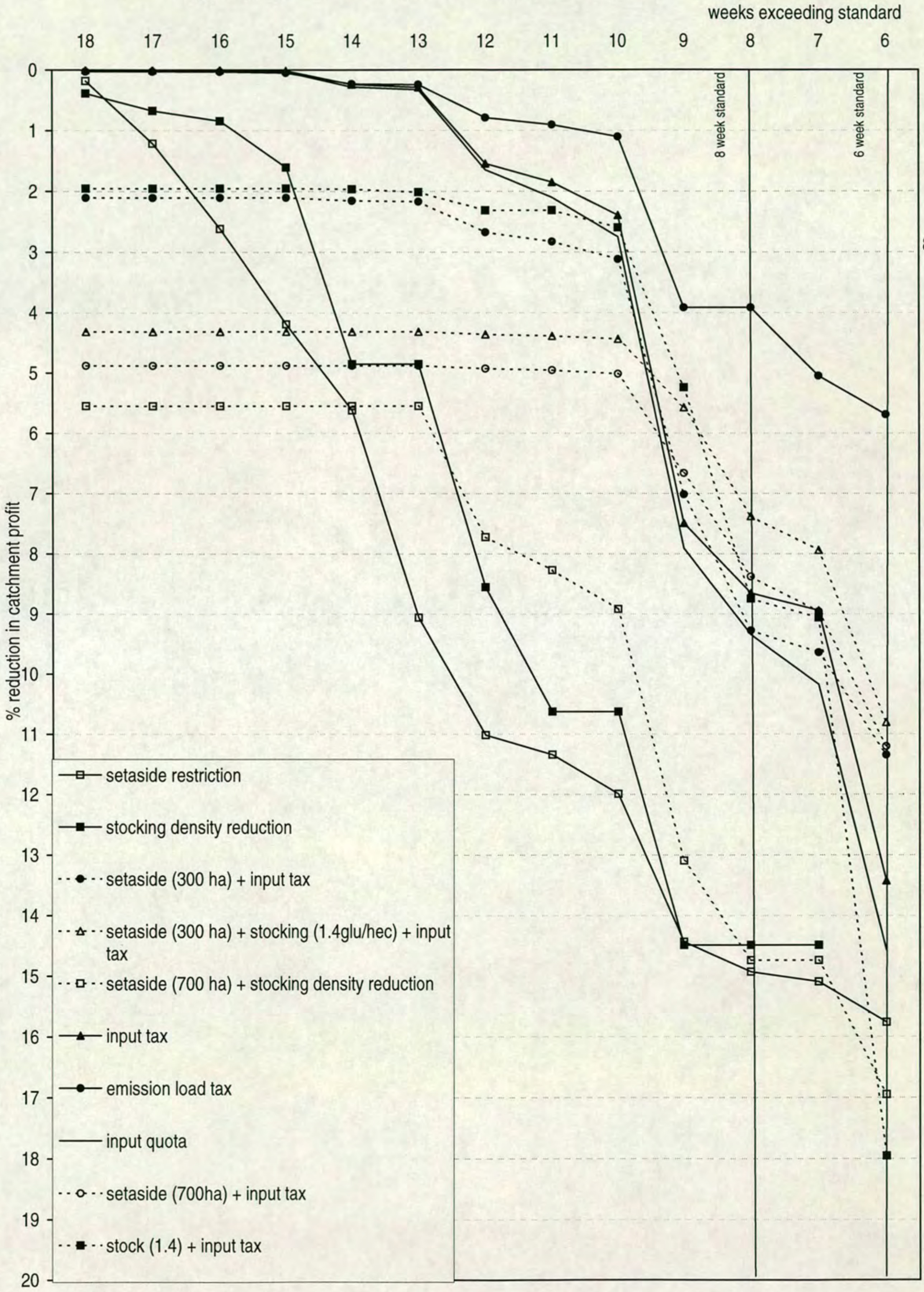


Table 7.13: Instrument Levels with Different River Flow Restrictions and Weather Conditions

Regulatory Target	Standard should not be Exceeded more than 8 Weeks			Standard should not be Exceeded more than 6 Weeks		
Weather	Mean Year Rank		Wet Year Rank	Mean Year Rank		Wet Year Rank
River Flow Restriction	90th %ile river flow restriction	No river flow restriction	With or without any river flow restrictions	90th %ile river flow restriction	No river flow restriction	With or without any river flow restrictions
Emission Tax (per kg of emission)	4.93	5.59	11.24	5.92	7.25	12.35
Input Tax (required nitrogen price (£/kg))	1.4	1.54	5.7	1.62	1.97	13.95
Setaside (300 ha) + Input Tax (£/kg))	0.98	1.2	4.32	1.15	1.51	7.5
Setaside (300 ha) + stocking density reduction (1.4 glu/ha) + input tax (£/kg)	0.6	0.65	2.31	0.69	0.77	5.6
Setaside (700 ha) + stocking density reduction (glu/ha)	1.2	1.1	0.7	1.1	1	0.6
Stocking density (1.4 glu/ha) + input tax	0.77	0.87	4.12	0.91	1.05	25.6

Stocking density reduction re-allocates land from arable crops to grassland (less polluting) resulting in considerably reduced nitrogen application per hectare to grassland. Although still comparatively *not cost-effective*, stocking density reduction performs *relatively* better in a wet year than input taxation and quota (at least at the 8 week standard compliance target).

Although estimated emission taxation retains the highest rank, emission based policies are known to have high monitoring costs (Falconer 1998; M.A.F.F. 1998) and other aforementioned problems[85] (chapter 3), the regulator might turn to other policies including mixed instruments. In comparison the transaction costs associated

[85] EBPs are subject to uncertainty due to 'background nitrogen emissions', resulting from the natural variability of soils, variability in present crop uptake and effect of previous crop rotations (Lord 2001a).

with setaside, livestock density reduction and input taxation are arguably lower (Kampas and White 2002) . The cost of input taxation is in the range of 0.5 – 1% of total cost (OECD 2001), while it is likely that the cost of enforcing stocking density reduction has fallen in the aftermath of the BSE and foot and mouth crisis due to improved data collection. Similarly assuming setaside land is permanent, which is the most environmentally sensible option in terms of leaching, lowers the cost of monitoring considerably.

A three instrument mix of setaside (300ha), stocking density (1.4) and input taxation was the second most efficient policy after emission based policies. However this level of the three instrument policy is not as cost-effective in the mean year where stocking (1.8 glu/ha)/setaside (300ha)/input taxation ensures compliance at least cost. Similarly the improvement from a setaside (700ha)/input tax mix in the wet year is not borne in a year with 'mean' weather where it results in nearly an 8% reduction in catchment profitability. In a 'mean' year setaside (300ha)/input tax policy ensures compliance with only a 3% reduction in catchment profit. Overall, considering both setaside levels in conjunction with an input tax the setaside (300ha)/input tax is superior as the setaside (700ha)/ input tax offers only marginal improvement under wet weather conditions, relative to its costliness in mean weather conditions.

Stocking density reduction/input taxation mix (at both the 1.4 and 1.8 glu/ha level), which does well in the mean year, offered no improvement on setaside mix policies in the wet year and remained slightly less cost-effective than input taxation. A stocking density of 1.8(glu/ha)/input taxation was also considered but proved particularly costly in the wet year (not shown in figure 9). Overall this implies that there is *a cross-over in the relative efficiency ranking of policies between weather scenarios.*

Thus the policy ranking listed earlier in table 7.12 is an oversimplification of the regulator's problem because it compares policies based on their optimal levels in each weather and river flow restriction scenario. For example the three instrument policy was tested at both the 1.8 and 1.4 (glu/ha) stocking density restriction for all weather conditions and river flow restrictions. Of the two policy levels the one which performs best in each weather and river flow scenarios was considered in table 7.12.

Emissions were taxed incrementally and subtracted from the objective function (Dosi and Moretto 1994; Shortle and Abler 1994). The efficiency of such taxation can be seen by the widening gap between emission taxation (at the 6 week target) and all other polices from figure 7, through figure 8 to figure 9 which in effect reflects represents increased water availability. The *efficiency of estimated emission taxation in the model is misleading* as it assumes a) the farmer perfectly comprehends the regulator's modelled relationship between his management practices, application of nitrogen, catchment weather patterns and emissions, i.e. the bio-physical simulation model b) the farmer is risk neutral (Schmutzler 1996) c) farmer has the same expectation of weather conditions as the regulator (Shortle and Dunn 1986)[86]. In practice models at present cannot provide emission estimates accurately enough to withstand legal challenges and the costs of running complex models can be prohibitively high (Weersink et al. 1998). Additionally there are issues with the political acceptability of the information required to run an estimated emission model (Chambers 1992). It is likely that if an emission tax is levied it will require input and management monitoring besides soil testing on farm fields – a very expensive process.

If the regulator opts for an *input based policy* the farmer equates the cost of nitrogen application with his marginal benefit; thus his expectation of future weather nor his risk preference affect his nitrogen application decision. Thus it is more likely that farmers will behave in accordance with the regulator's desired behaviour.

Ultimately the regulator faces the difficult decision of choosing a policy which does reasonably well in *all weather conditions* not just mean conditions. As nitrate loadings are subject to a high degree of variability both within and among different years (Halstead et al. 1991), i.e. weather is stochastic, the regulator's decision should

[86] The authors assume that farmer and agency have identical information structures for weather.

be based *not only on the expected weather but also its variance*[87] (Braden and Segerson 1993; Shortle and Abler 1997). Another important consideration is the required level of standard compliance (regulatory target) and *regulator's risk aversion to the level of standard compliance being exceeded in extremely wet years.* If the regulator is particularly risk averse and works under the precautionary principle he/she may set a policy level based on the worst case wet weather scenario. *The greater the risk aversion to the standard being exceeded the more likely the regulator will favour policies which perform better in wet years.* This might offer another explanation for regulators reluctance to implement economic instruments (Hanley et al. 1990). Given wetter winter conditions are likely to prevail in Scotland under *climate change* (Hulme and Jenkins 1998; Kerr et al. 1999; Harrison et al. 2001) it is possible that optimal diffuse nitrogen regulation in the future will be similar to what is optimal in the extreme 'wet year'.

Of course categorising weather into representative categories is a simplification for illustrative purposes and in reality there can be a great deal of variation between weather conditions. In practice the regulatory level setting will involve a lot of trial and error and depend on catchment specific characteristics and the wider regulatory objective. While the availability of realistic transaction cost estimates would result in better policy ranking based on social costs, the possibility of targeted policies (Kolstad 1987; Shortle and Horan 2001) conferring improvements needs to be investigated.

7.6.1 Conclusion

This study researched the efficient management of two agricultural externalities, i.e. diffuse nitrogen pollution in rivers and low river flows from surface water extraction for irrigation. It found that as a means to control diffuse nitrogen pollution imposing river flow controls were not cost-effective. However since MAF river restrictions contribute towards pollution mitigation they should be accounted for in the design of nitrogen control policies as they reduce compliance costs.

[87] i.e. the mean and variation of emission concentrations between years.

Secondly the study ranked policies to control diffuse nitrogen pollution in terms of their abatement cost under mean and wet weather conditions. The ranking of control measures under mean weather conditions conformed, for the most part, to the economic literature hierarchy. However in comparison under wet weather conditions the cost-effectiveness of economic instruments is significantly reduced. Control policies may be set based on wet weather conditions if the regulator is extremely risk averse to the standard being exceeded. Wet weather conditions might prevail in the future given climate change.

Although the efficiency of mixed instrument policies has been examined in the literature (Braden and Segerson 1993; Millock and Salanie 1997; Schmutzler and Goulder 1997) nearly all published research considers only 2 instrument mixes of economic instruments alone. This study investigated four mixed instrument policy packages combining *economic incentives and non-economic (managerial) approaches* 1) setaside reduction and input taxation, 2) setaside and stocking density reduction, and 3) stocking density and input taxation 4) setaside and stocking density reduction with input taxation were examined. All four were considered with and without the 90th percentile river flow restriction. There is no known mention of a three instrument mix policy in the literature. One reason for this may be that the introduction of an additional instrument results in additional and prohibitive transaction costs, while this is probably true of combining complex economic instruments, non-economic or managerial regulatory approaches (setaside land and stocking density reduction) are arguably cheaper to enforce. An argument in favour is the presence of IACS data collection and existing stocking density and setaside restrictions which are currently enforced under the AAPS and other schemes.

Chapter 8
Limitations and Further Research

8.1 Introduction

The study limitations as well further improvements in research methodology are briefly presented in the initial segment of this chapter. This is followed by a discussion on the political economy of environmental regulation and why economically efficient solutions are not necessary political ones. The last section of the chapter is a thesis summary stating the main research objectives, originality, results and their practical significance.

8.2 Research Limitations and Improvement

An obvious limitation is the partial optimisation framework of the model. Labour and capital costs were not considered in modelling farmer production decisions. This can result in erroneous crop mix estimates and change the relative cost of alternative NPS nitrate control policies. In theory, labour and capital costs can be estimated, but this would require either detailed co-operation from actual farmers in the catchment or assumptions regarding estimates of labour (such as the value of family labour etc.) and farm capital costs. The study also ignores the accumulation of nitrate, and overestimates compliance costs because the model involves a static optimisation framework.

The absence of transaction costs in the analysis is a considerable limitation. Arguably this is the most consistently overlooked issue in determining the cost-effective ranking of NPS nitrate control policies in the literature. Although their absence is recognised repeatedly in the literature, they are for the most part conveniently ignored casting doubt on the validity of relative instrument ranking. The need for realistic transaction costs (including those of targeted policies) estimates should definitely be a research priority.

Ambient taxation was modelled as a levy on positive deviations from the regulatory target river ambient concentration. However the results were not presented because strictly speaking this is not 'ambient taxation' as proposed by Segerson (see chapter 3). A correctly specified Segerson ambient based regulatory policy involves both a tax and subsidy depending on whether ambient levels are positively or negatively deviated. However this could not be tested because the available solvers could not deal with mixed integer non-linear programming problems.

Overall more realistic rotational constraints which a) incorporate all crops, not just the major ones, and b) allows potatoes to be grown on all soil textures, not just a sandy texture, will allow more robust emission estimates. Unfortunately nitrate leaching data on 'minor' crops were not available. Similarly if more detailed livestock statistics had been available it would have permitted more realistic instrument levels and possibly even ranking.

The availability of detailed hydrological/geological catchment modelling would have meant not having to make assumptions regarding the ground water component of river flow and improved the reliability of model results. Regarding the transferability of research to other catchments it should be noted that the impact of river flow controls on diffuse pollution depends on *catchment specific* MAF restrictions, regulatory targets, farming's dependence on surface irrigation water, weather paterns, catchment hydrology and spatial aspects of extraction. In designing diffuse nitrogen pollution regulation in the presence of river flow restrictions the income distributional effects on irrigating and non-irrigating farmers will also need to be considered. Unfortunately the model assumed that potatoes were only grown on sandy soils. It would be interesting to compare the efficiency gains from river flow controls on potato leaching on other soil types.

The result that mixed instruments confer benefits in practice is not surprising; the pervasive use of multiple instruments is well documented in the literature.

'*It is unreasonable to think that a single instrument is likely to be suited to the myriad of tasks involved in implementing an environmental policy...the use of multiple instruments will tend to be the rule rather than the exception.*' (Hahn 1990).

The bio-physical model's sensitivity to certain key parameters and constraints must be noted. Obviously the crops included in the rotational constraints and their duration is very important, both in terms of profitability and nitrate emissions. The model is sensitive to the upper bounds on a) the potato crop (under all irrigation regimes) acreage, and b) dairy cows (GLU) numbers . This is primarily because both activities are highly profitable but also polluting. Sensitivity analysis revealed that crop price is a key factor influencing model predictions . A simulated crop price decrease resulted in significant reduction in the nitrate pollution generated - as is theoretically expected. However, the sensitivity analysis carried out was not extensive and only done for 15% and 20% decrease in the price of potatoes, spring barley and winter wheat. The grass utilisation coefficient, which is the proportion of grass grown utilised by grazing animals appears to be another important determinant of model results and instrument ranking.

8.3 Political Economy of Environmental Regulation

A positive analysis of actual agri-environmental policy choice reveals instrument use is not determined by economic efficiency alone. Additional criteria include equity and political considerations. The opposition to market based instruments by farmers stems from the additional costs they bear under such regulation, which may force them out of business in the long run (Dewees 1983). Hence it is reasonable to expect that affected interest groups co-ordinate their resources to lobby against the adoption of market based incentives (Daugbjerg 1998).

Thus economic agents (farmers) are motivated to influence environmental policy as instrument choice (emission standards over emission taxes) has income distributional consequences (Buchanan and Tullok 1975). Emission standards serve as an entry barrier to new firms, thus raising the profits of existing firms - whereas taxation does not preclude new entry.

Some models of 'political internalisation' exist whereby environmental policy is the outcome of political self-interest and where 'political competition' is a source of the internalisation of economic instruments (Aidt 1998). Under the Coasian tradition, affected parties mobilise to protect their interests via political markets and not through a private transfer scheme - which is presumably more costly. A self-interested policy maker with coercive power to implement policies trades-off the demands of lobby groups against the general interest of voters. If all agents have their interests represented by lobby groups then the political equilibrium is socially efficient. However in reality some groups face lower organizational cost and overcome the free rider problem of collective action – thereby coordinating better than others (Aidt 1998).

The issue of pollution taxation versus direct regulation can be considered a disagreement between government and producer-interests. The government or regulator prefers taxation as enforcing direct control requires high enforcement costs. Whereas the lobby prefers direct regulation since it is cheaper and serves as a coordination device to create short-run monopoly rents. The dispute is resolved by arguing that small influential and organised producers are more efficient at lobbying than the larger public - which presumably supports taxation since it generates revenues. Thus at political equilibrium direct controls are chosen despite the efficiency of taxation.

Many influential interests group have their own criteria and hence society has varied preferences regarding the choice of regulatory control. The revealed political preference function can be thought of as a weighted sum of individual interest groups' preference measures (Gardner 1987; Hahn 1989b) - where groups exerting more political clout have a higher weight. A simple preference function would include consumer and producer surplus, net government revenue, and environmental costs. If all of these are equally weighted then the political preference is for more efficient policies; however for both agricultural and environmental policy it is well known that they are not (Gardner 1987; Bromley and Hodge 1990).

Influential interest groups include farm organisations (land owners), chemical input suppliers (fertiliser manufacturers) and environmental groups (general public). Most environmental groups favour direct regulation over incentive based economic instruments. They mistrust the effectiveness of economic incentives and believe market based policies are simply 'rights to pollute' as they fail to stigmatise pollution as inherently wrong (Kelman 1983). Another reason sited for environmental groups' preference for direct regulation include their apparent involvement in determining methods and levels of abatement under command and control regulation (Bohm and Russell 1985). Interestingly over time polemic views on environmental taxation have been discarded. To illustrate, many leading environmental groups such as Greenpeace International are calling for extremely high progressive taxes to phase out nitrogen and pesticide use (Clunies-Ross 1993).

An additional factor against the adoption of economic controls in practice is that until recently in some EU countries and the US, bureaucratic preferences have been complicated by the conflicting interests of agricultural and environmental agencies (Shortle and Abler 1991). Where environmental policies for agriculture are administrated by agricultural agencies there is an interest in maintaining the current system of regulation and support from farmers. In practice political acceptability is also determined by government budgets, administration and enforcement costs; higher transaction costs reduce political acceptability.

Unfortunately, for the most part, traditionally the primary purpose of regulation is not to limit pollution to acceptable levels. More often than not they are used to raise revenue for public pollution control activities, research and development etc. Thus in designing and implementing agri-environmental policy a successful outcome is not measured by efficient achievement of pollution control alone.

8.4 Research Summary

This study *theoretically* modelled the first and second-best internalisation of two surface water externalities in an agricultural catchment. The two agricultural externalities considered were *non-point source nitrogen pollution* and *reduced river* *flows from surface water irrigation*. It introduced the concept of *complimentary interaction* between controls which target different agricultural externalities, i.e. surface water NPS nitrate pollution (quality) and low river flows (quantity).

To date, numerous aspects of diffuse nitrogen pollution as a negative production externality have been examined. The research also *theoretically analysed the efficiency gains from recognising the dual nature of surface water diffuse nitrogen pollution* both as a *positive and negative externality* in a catchment where the regulator wishes to enforce a minimum river flow and ambient water quality standard. There is no known analysis of NPS control instruments which recognises the dual nature of nitrogen in the published literature.

Empirically this study researched the efficient management of two agricultural externalities, i.e. diffuse nitrogen pollution (as a negative externality only) and low river flows from surface water extraction for irrigation in a Scottish catchment. It found that as a means to control diffuse nitrogen pollution imposing river flow controls were *not cost-effective*. However since MAF river restrictions contribute towards pollution mitigation they should be accounted for in the design of nitrogen control policies as they reduce compliance costs.

Although the efficiency of mixed instrument policies has been examined in the literature nearly all published research considers only 2 instrument mixes of economic instruments alone. This study investigated four mixed instrument policy packages combining *economic incentives and non-economic (managerial) approaches*, i.e. 1) setaside reduction and input taxation, 2) setaside and stocking density reduction, and 3) stocking density and input taxation 4) setaside and stocking density reduction with input taxation. All four were considered with and without the 90^{th} percentile river flow restriction.

There is no known mention of a three instrument mix policy in the literature. One reason for this may be that the introduction of an additional instrument results in additional and prohibitive transaction costs. While this is probably true of combining complex economic instruments, non-economic or managerial regulatory approaches (setaside land and stocking density reduction) are arguably cheaper to enforce and are already currently enforced under other agricultural support schemes.

The study also ranked policies to control diffuse nitrogen pollution in terms of their abatement cost under *dry, mean and wet weather conditions*. Through repeated runs, a *continuous spectrum of instrument ranking* from the baseline to different arbitrary regulatory targets was achieved. Continuous ranking enabled the identification of 'cross-overs' in efficiency ranking. The ranking of control measures under mean weather conditions conformed, for the most part, to the economic literature hierarchy. However, in comparison *under wet weather conditions the cost-effectiveness of economic instruments relative to mixed instruments is significantly reduced*. In fact certain mixed instrument control policies outperformed conventional input taxation and quotas. Control policies may be set based on wet weather conditions if the regulator is extremely averse to the regulatory target (required standard compliance) being exceeded. Wet weather conditions might prevail in the future given climate change.

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There is an abundance of debate surrounding pigouvian taxation (e.g. entry/exit incentives) and related theoretical issues such as 'prices vs quantities':

1. Pigouvian taxation is inefficient because the total payments in tax exceed the total external damages, resulting in suboptimal entry conditions (Rose-Ackerman 1973; Schultze and d'Arge 1974; Collinge and Oates 1982). Put differently, even in a perfect scenario, an optimal pollution tax will over penalise as even when operating at the optimal efficiency level, where marginal externality cost and marginal private benefit curve cross, the polluter will still be paying a tax (Pezzy and Park 1998).
2. Pigouvian taxation is inefficient when the marginal external cost function slopes upwards (Burrows 1979).
3. The correction of the above inefficiency (point 1) through the use of lump-sum transfer payments (Carlton and Loury 1980; Carlton and Loury 1986).
4. Assuming the standard 'perfect competition' assumption the contribution of an individual firm is 'negligible', thus the marginal social costs are approximately constant over a single firm's emissions (Baumol and Oates 1988).
5. Within a general equilibrium framework, the fact that pigouvian taxes exceed total external damages is a 'normal' phenomena akin to consumer or producer surplus (Kohn 1985; Kohn 1994).
6. The impact of pigouvian taxation on industry entry/exit conditions depends on the elasticity of demand. If demand is elastic then entry/exit will be affected, but not if demand is inelastic (Conrad and Wang 1993).
7. Most analysis is simplified by a continuously increasing marginal abatement cost function (MAC), implying a local and global cost minimisation solution that coincides. Rowley et al., 1979, undertook a simulation model of point source discharges on the Tees, to determine whether the theoretical cost minimisation of pollution taxes over standards could be shown to exist and found that economies of scale were present. The aggregate abatement cost function was not strictly convex and any local maximum under a tax scheme was not necessarily a global maximum. As discharges along a river's length differ in output, age, design, technology, location, production processes, managerial skills etc., their MAC functions should also reflect this variability. Hanley and Moffat, 1993, found that MACs for direct discharges of BOD to the Forth Estuary in Scotland differ as much as thirty times.
8. When competitive economic conditions are satisfied efficiency will not require any extra entry/exit conditions (Xepapadeas 1997). Some have argued that marginal conditions are left unaffected in the long run (Migue and Marceau 1993).
9. Regarding the choice of price or quantity instruments, Weitzman concluded that the relative slopes of the relative cost and benefit curves matter and uncertainty regarding costs is the most important consideration. He argued that quantity instruments achieve a lower expected welfare loss than price

instruments if the marginal benefits curve is flatter than the marginal cost curve and vice versa (Weitzman 1974). Essentially Weitzman showed that if the regulator has imperfect information regarding *MAC* then he will be able to achieve the desired emission reduction using a standard but may greatly over or underestimate the cost of achieving it. If he knows the marginal costs of firms and employs an emission fee he will be uncertain of the level of emission reduction.

10. Later work concluded that choice of instrument is contingent upon the degree of cost uncertainty and the relative slopes of both curves – thus a generalisation is not possible (Yohe 1978). This conclusion was confirmed for when firms are risk averse (Adar and Griffin 1976) and also in the presence of price distortions (Schob 1996). The effect of correlated uncertainty regarding both the benefit and cost curves concluded that a positive correlation tends to favours quantity controls, whereas negative correlation favours price intervention (Stavins 1996). With agricultural pollution such statistical dependence would be because of weather which influences both yield and emissions.
11. In comparing the relative efficiency of effluent and input taxes the return to scale of the leaching function is an important factor (Stevens 1988). If the leaching function is characterised by increasing returns to scale, i.e. the degree of homogeneity is greater than 1, then an effluent tax will be more efficient. However, if the leaching function exhibits decreasing returns to scale (degree of homogeneity less than 1) than an input tax will be more efficient. Here efficiency implies reducing farmer income loss, as measured by money/kg of N reduced. The leaching function varies depending on site-specific characteristics, thus both an effluent and input tax must be site-specific to be efficient, gathering site-specific information is very costly. Thus if this is not done and a uniform tax is applied then there will be too much reduction in areas where there is little emissions and too little where there are considerable emissions.